

**The ecohydrology and conservation of a
coastal sedimentary lake and wetland system:
Sheskinmore Lough, Donegal, Ireland**

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Thesis submitted for the degree of
Doctor of Philosophy (PhD)

June 2016

I, Elizabeth Ann Gardner, confirm that the work presented in this thesis is my own. Where information has been derived from other sources, I confirm that this has been indicated in the thesis.

21st June 2016

Abstract

Lowland, shallow, coastal lake systems often comprise a complex array of habitats and species as a consequence of their geomorphic evolution, in combination with marine and terrestrial forcing. But they are also vulnerable to changes in climate and human activities that both influence species assemblages, sediment dynamics, water quality and hydrology. Sheskinmore Lough, located on the west coast of Donegal, northwest Ireland comprises a shallow (<1.5m) freshwater sedimentary lake surrounded by a diverse array of coastal and freshwater wetland and dune habitats supporting a plethora of rare and endangered species. The lake-wetland-dune complex, designated under the EU Habitats and Birds Directives, is managed by the National Parks and Wildlife Service (NPWS) who are concerned that declining water levels are driving negative impacts on protected flora and fauna; however their water management approach is reactionary, lacks an ecohydrological basis and is inherently unsustainable. The aim of this PhD is to inform conservation management strategies via multidisciplinary analysis of the ecohydrology of Sheskinmore Lough and its adjacent wetlands. The thesis examines the contemporary ecohydrology of the system, and reconstructs past environmental change using multiproxy paleolimnological techniques to ascertain the envelope of ecohydrological variability over different timescales. In addition, the research uses a distributed hydrological model to explore the impacts associated with climate change and hydrological management.

The results reveal a lake and wetland system that has a complex contemporary ecohydrology set in a complicated coastal environment. Ecological analysis indicates an oligotrophic, circumneutral, shallow lake system, overlying a sedimentary complex dominated by peat and calcareous sandy substrates, fringed by a wetland system comprising fen and mire communities that also favour similar conditions. Hydrology was identified as a key factor influencing the distribution and composition of communities across the site. Operation of the sluice had the greatest impact, causing water levels to fluctuate rapidly (up to 1m in under 7 days) within the lake, with knock-on effects observed across a large part of the wetland system. Paleolimnological analyses revealed two important climatic and geomorphological shifts defining three key phases in the recent environmental and ecohydrological history of the site. First, a change occurred in the mid to late 1500s AD from a drier, sandy environment when the lake was primarily a riverine system, to one that was wetter and dominated by peat and reedbeds. The second transition occurred c.1800 AD when the climate became more turbulent, prompting the development of a lake-wetland system. Finally, modelling projections indicate the lake and wetland system are likely to experience increasing impacts in the future due to a more variable climate and lake water levels fluctuating more as a result. Ultimately, hydrological management coupled with climate change presents the greatest potential ecological threat to Sheskinmore Lough. This thesis therefore provides a series of conservation recommendations to enhance the preservation of similar freshwater systems, while the knowledge gained contributes significantly to the understanding of shallow aquatic ecohydrology.

Contents

Declaration	3
Abstract.....	5
Contents	7
List of figures	11
List of tables	23
Acknowledgements	27
1 Introduction	29
1.1 Aquatic environments	29
1.2 Wetland ecosystems.....	30
1.2.1 Wetland definitions and classifications.....	31
1.2.2 Wetland extent and distribution	36
1.2.3 Threats to wetlands	37
1.2.4 Wetland conservation management.....	39
1.3 Lake ecosystems	41
1.3.1 Lake classification	41
1.3.1.1 Macroscale: centuries to millennia	42
1.3.1.2 Mesoscale: decades to centuries.....	44
1.3.1.3 Microscale: months to years.....	46
1.3.2 Lake ecology	48
1.3.3 Lake hydrology.....	51
1.3.4 Lake water level regime	53
1.4 Lake and wetland ecohydrology	55
1.5 Coastal freshwater lake-wetland systems	57
1.6 Research objectives	58
2 Study area	61
2.1 Northwest Ireland: ecophysical context.....	61
2.2 The coastal lakes of northwest Ireland.....	63
2.3 Sheskinmore Lough	66
2.3.1 Hydrogeomorphological development.....	67
2.3.2 Ecology and conservation.....	69
3 Methodology	75
3.1 Research design	75
3.2 Ecology	76
3.3 Hydrology.....	79
3.3.1 Surface and groundwater hydrological monitoring.....	79

3.3.2 Meteorological survey.....	86
3.3.3 Water sampling.....	88
3.3.4 Flow gauging.....	89
3.3.4 Topographical survey.....	95
3.4 Paleolimnology.....	96
3.4.1 Exploratory cores.....	97
3.4.1.1 <i>Geochemistry</i>	97
3.4.1.2 <i>Particle size analysis</i>	97
3.4.1.3 <i>Macrofossils</i>	99
3.4.1.4 <i>Diatoms</i>	100
3.4.2 Lake cores.....	101
4 Contemporary ecohydrology of Sheskinmore Lough.....	107
4.1 Introduction.....	107
4.2 Ecology.....	107
4.2.1 Wetland and reedbed macrophytes.....	108
4.2.2 Lake macrophytes.....	114
4.2.3 Ecological classification of the lake and wetland system.....	120
4.3 Edaphic environment.....	130
4.3.1 Soil and water chemistry.....	130
4.4 Hydrology.....	132
4.4.1 Temporal scales and patterns in local climatology.....	132
4.4.2 Lake and wetland hydrology.....	139
4.4.2.1 <i>Two-year hydrological survey: seasonal trends</i>	143
4.4.2.2 <i>Six-week hydrological survey: the role of precipitation</i>	149
4.4.2.3 <i>Four-week hydrological survey: the role of the sluice</i>	153
4.4.2.4 <i>Seven-day hydrological survey: spatial connectivity of the system</i>	155
4.4.3 Hydrological classification of the lake and wetland system.....	161
4.5 Discussion.....	170
4.5.1 Wetland community-environment relationships.....	170
4.5.2 Hydrological requirements of communities and habitats.....	178
4.5.3 Conceptual model of site ecohydrology.....	187
4.6 Conclusions.....	189
5 Multi-proxy paleolimnology of Sheskinmore Lough.....	191
5.1 Introduction.....	191
5.2 Wetland and lake sediments and stratigraphy.....	191
5.2.1 Sedimentology.....	192
5.2.2 Paleoecological indicators.....	194
5.2.3 Summary.....	198
5.3 Lake sediment stratigraphy.....	198
5.3.1 Lithostratigraphy.....	198
5.3.2 Palaeoecology.....	204

5.3.2.1 <i>Macrofossils</i>	204
5.3.2.2 <i>Diatoms</i>	208
5.3.3 Stratigraphic evaluation	212
5.3.4 Core chronology	215
5.3.5 Historical context	218
5.4 Discussion	224
5.5 Conclusions	232
6 Modelling the future ecohydrology of Sheskinmore Lough	235
6.1 Introduction	235
6.2 Hydrological modelling	235
6.2.1 Introduction to hydrological modelling	235
6.2.2 The MIKE SHE modelling system	238
6.3 The MIKE SHE model of Sheskinmore Lough	241
6.3.1 Model development	241
6.3.2 Model calibration and validation	254
6.4 Climate change impacts	261
6.4.1 Development of climate change scenarios	262
6.4.2 Modelled hydrological response to climate change	263
6.5 Hydrological management impacts	272
6.5.1 Selection of management scenarios	272
6.5.2 Modelled hydrological response to management decisions	272
6.5.3 Modelled hydrological response to management decisions and climate change	274
6.6 Discussion	278
6.6.1 Ecological responses to climate change	278
6.6.2 Ecological responses to hydrological management	281
6.6.3 Relative impacts of climate change and management on system ecohydrology	287
6.7 Conclusions	289
7 Conclusions and recommendations	293
7.1 Introduction	293
7.2 The ecohydrology of Sheskinmore Lough: past, present and future ...	293
7.3 Contributions to coastal freshwater science	297
7.4 Recommendations	298
7.4.1 Conservation management	298
7.4.2 Further research	300
References	303
Appendix	359

List of figures

Figure 1.1 Environmental factors determining wetland community and ecosystem type (adapted from Keddy, 2010; p31).	31
Figure 1.2 Wetland classification based on two sets of environmental factors: water regime and nutrient supply (adapted from Gopal et al, 1990; cited in Keddy, 2010; p36).	34
Figure 1.3 Classification tree of wetlands and deepwater habitats (sourced from Cowardin et al, 1979; p5).	35
Figure 1.4 Conceptual model showing wetlands at the centre of concerns of society (adapted from Maltby & Acreman, 2011; p1342).	40
Figure 1.5 Lake macrophyte and ecological zonation including the open water euphotic zone and the benthic zone. Note that the benthic zone representation has been simplified as it actually extends all the way to the emergent zone at the lakeshore.	49
Figure 1.6 Examples of aquatic macrophytes: (a) Emergent (<i>Phragmites australis</i>); (b) Submerged (<i>Chara aspera</i>); (c) Free-floating (<i>Lemna minor</i>); (d) Floating-leaved (<i>Potamogeton natans</i>).	50
Figure 1.7 Flow diagram showing the key hydrological components associated with lake water budget.	52
Figure 1.8 The integration of hydrology and ecosystem science - ecohydrology - in water management (adapted from Nuttle, 2002: 806).	56
Figure 2.1 The hydrology and topography of Ireland (adapted from NASA, 2000).....	62
Figure 2.2 Conceptual model showing coastal lake community types due to salinity and other factors including tidal influence, isolation from marine processes and sedimentation (Adapted from Oliver, 2007, p25).	66
Figure 2.3 Sheskinmore Lough SPA, its location in County Donegal and Ireland, and the position of the lake, the inflowing Abberachrin and Duvoge rivers and the outflow relative to the Loughros More estuary (Bing, 2012).	67
Figure 2.9 The channel feature in Sheskinmore Lough highlighted within the 2012 satellite image (Bing, 2012).	69
Figure 2.5 Selection of photographs of the Sheskinmore Lough lake system: a) the western half of the lake with reedbed and wetland beyond, viewed from the southwest; b) the northeastern half of the lake with the Duvoge river mouth beyond, viewed from the south; c) the wetland at the southern side of the lake during high water level conditions, viewed from the south; d) the reedbed fringe along the northern side of the lake with uplands beyond, viewed from the southeast; e) the reedbed and Duvoge river, viewed from atop the cliffs on the eastern side of the lake; f) the reedbed, lake and mouth of the Duvoge river, viewed from atop the cliffs on the eastern side of the lake; g) the cliffs on the eastern side of the lake by the Abberachrin river mouth, viewed from the south; and h) the outflow looking downstream towards the estuary via the sluice and dune system, viewed looking south.	70
Figure 2.6 Selection of photographs of the Sheskinmore Lough wetland, dune and surrounding systems: a) the Duvoge river floodplain viewed looking downstream towards the lake from the northeast, with reedbed and wetland beyond; b) ponding and the predator fence within the reedbed, viewed looking north; c) cattle	

grazing the wetland along the southern side of the lake, viewed from the northwest; d) the central and western arm of the wetland system and estuary beyond, viewed from the uplands to the north; e) the western end of the wetland, viewed from the southeast; f) the central wetland system and dunes beyond, viewed looking south; g) the large blowout within the dune system near to the western end of the wetland, viewed looking southeast; and h) one of the ponds on the beach at Tramore Strand at the base of the dune system, viewed looking east.
.....71

Figure 3.1 Location of the 90 wetland quadrat sites surveyed in June 2012 and June 2013 and also the 33 reedbed quadrat sites surveyed along transects in July 2014 in relation to the total habitat areas. Note that, although some quadrats lie outside the reedbed area delineated from the 2012 satellite image, they are all located within the 2014 reedbed area.....77

Figure 3.2 Location of the 65 lake sample sites surveyed in June 2012 and June 2013, in relation to the total open water habitat area and the 4 transects located across the highlighted channel feature and the littoral zones surveyed in July 2014.
.....79

Figure 3.3 The main hydrological features and the location of hydrological monitoring sites at Sheskinmore Lough. Sites monitoring the lakes and river systems (L1-L6) are shown in blue; sites monitoring the wetland (W1-W7) are shown in red; sites monitoring the surrounding areas (S1-S5) are shown in orange; sites monitoring the dune system (D1 & D2) are shown in yellow; and the weather station (WSt) is shown in purple. Circular symbols indicate dipwell groundwater monitoring sites; square symbols indicate surface water monitoring sites; and the pentagon symbol indicates the weather station.....80

Figure 3.4 The sluice gate and pedestrian bridge structure observed from two angles at low flow levels in July 2013. The sluice gate comprises a movable gate driven by a hand-operated rack-and-pinion drive mechanism that allows water to flow under it when the sluice is raised. At these times the sluice is 'open'. At times of very high flow when the sluice is 'closed' and no water can pass underneath, water may spill over the top and at these times the sluice gate operates as a weir. Note the rectangular concrete structure with metal grill on the top to the right of the sluice gate houses the fish pass.81

Figure 3.5 Diagram a) and photograph b) showing the groundwater logger apparatus (dipwell and suspended water logger) and the downloading of a TROLL logger.
.....82

Figure 3.6 Diagram a) and photograph b) showing the surface water logger apparatus (concrete and tube) in the lake and inflows.83

Figure 3.7 Diagram a) and photograph b) showing the surface water logger apparatus (wooden stake and tube) in the dune ponds.83

Figure 3.8 Diagram a) and photograph b) showing the surface water logger apparatus (tube and suspended logger) attached to the sluice gate structure.
.....83

Figure 3.9 Photograph of the sealed top of a dipwell within the wetland.....84

Figure 3.10 Location of the hydrological monitoring sites for the year-long survey (all coloured symbols except L7) and also for the discrete monitoring periods (all symbols). Sites monitoring the lakes and river systems (L1-L7) are shown in blue; sites monitoring the wetland (W1-W7) are shown in red; sites monitoring the surrounding areas (S1-S5) are shown in orange; sites monitoring the dune system (D1-D2) are shown in yellow; and the weather station (WSt) is shown in purple.

	Circular symbols indicate dipwell groundwater monitoring sites; square symbols indicate surface water monitoring sites; and the pentagon symbol indicates the weather station. Sites not included in the year-long survey (L5, W4, S1, S2, S4 & S5) are shown as colourless.	86
Figure 3.11	The Davis Weatherlink VantagePro2 meteorological station installed in June 2012, locations of the Malin Head meteorological station and Sheskinmore Lough weather station in County Donegal and the monthly total precipitation and monthly average temperature at Malin Head downloaded from the Met Éireann website for the June 2012 to June 2013 period.	88
Figure 3.12	Calibration curves used to determine the slope of the curve (k) to calculate discharge in the a) Duvoige inflow (L1) and the b) Abberachrin inflow (L2).	92
Figure 3.13	The a) downstream electrical conductivity (EC) over time following upstream slug injection of the salt solution and the b) calibration curve used to determine the slope of the curve (k) and calculate discharge in the outflow (L4).	94
Figure 3.14	Points collected during the topographic and bathymetric survey conducted in June 2013 (Adapted from Maher-McWilliams, 2013).....	95
Figure 3.15	Locations of the exploratory sediment cores collected during the exploratory study in June 2012 from hydrological monitoring sites across the wetland system (S3, W1, W2, W3, W4, W6, W7) and from within Sheskinmore Lough (L3, L3.2, L3.6) and Sandfield Lough (L6).....	98
Figure 3.16	Location of the two full-length sediment cores collected from the channel feature within the lake in June 2013.....	103
Figure 3.17	Sheskinmore Lough Core 1 stratigraphy and selected radiocarbon samples (highlighted in red), their respective depths: A17 (12-13cm) & A67 (59-60cm) and their appearance within the core (see red marks in photographs).	106
Figure 4.1	Spatial distribution of habitat areas within Sheskinmore Lough SPA in 2012. The habitats were digitised and their areas calculated in ArcMap 10.1 using a combination of Bing (November 2011 to March 2012) satellite imagery, on-the-ground observations and GPS data (Bing, 2013).	109
Figure 4.2	Emergent macrophyte species richness and total abundance identified a) within the 90 quadrats surveyed in the wetland in June 2012 and 2013, and b) within the 33 quadrats surveyed in July 2014 in the reedbed habitat.	110
Figure 4.3	Distribution of a) wetland macrophyte species richness, b) reedbed macrophyte species richness, and c) percent cover of <i>Phragmites australis</i> at each sample site identified during the June 2012/2013 surveys and the July 2014 survey, respectively.	111
Figure 4.4	Expansion of the <i>Phragmites australis</i> reedbed from 2012 to 2014. The 2012 fringe was delineated from winter 2011- 2012 satellite imagery and a GPS survey was undertaken in July 2014 (Bing, 2012).	112
Figure 4.5	DCA's showing the relationships between the a) wetland macrophytes identified during the June 2012 and 2013 surveys, and b) reedbed macrophytes identified during the July 2014 surveys (<i>Phragmites australis</i> is highlighted in red). Samples are grouped based on their elevation (mOD). In a) 9 samples are located at 2.75-3.25mOD; 13 samples are located at 3.26-3.75mOD; and 11 samples are located at 3.76-4.25mOD. Eigenvalues: Axis 1 = 45.6%; Axis 2 = 17%. In b) 29 samples are located at 3.01-4.00mOD; 33 samples are located at 4.01-5.50mOD; and 28 samples are located at 5.51-7.50mOD. Eigenvalues: Axis 1 = 61.9%; Axis 2 = 15.9%.	113

Figure 4.6 Distribution of sample sites across the open water habitat of the lake identified during the June 2012 and 2013 surveys. The percent volume of lake water infested (PVI) with lake macrophytes was recorded at each sample site and is compared to elevation (mOD) via the DEM.	114
Figure 4.7 Lake macrophyte species richness and total abundance within samples sites in the open water habitat identified during the June 2012 and 2013 surveys.	115
Figure 4.8 DCA showing the relationships between the lake macrophytes species identified during the June 2012 and 2013 surveys. Samples are grouped based on average lake water depth (m) observed during the June 2012 to June 2013 monitoring period. Eigenvalues: Axis 1 = 74.9%; Axis 2 = 29.2%.	115
Figure 4.9 The locations of <i>Najas flexilis</i> (Slender Naiad) plants discovered in August 2013 and the zone of young plants observed along the channel feature within the open water habitat of the lake.	116
Figure 4.10 Macrophyte PVI within the open water habitat of the lake recorded along four transects located across the 'A' northern and 'B' southern sections of the channel (cross hatching), and two transects extending inwards from the 'C' southern and 'D' northeastern lake edge.	118
Figure 4.11 Schematic of submerged macrophyte species identified on 14th July 2014 along the two transects across the a) northern and b) southern channel feature and the two transects c) and d) towards the southern and north-eastern margins within the open water habitat of the lake in relation to average water depth (metres below the average water level).	119
Figure 4.12 Venn diagram detailing species found in the wetland and reedbed habitats, as well as those found in both habitats.	120
Figure 4.13 Dendrogram TWINSpan result for wetland and reedbed plant species data identified during the June 2012, 2013 and July 2014 surveys. Plant species in boxes represent indicator species for the point of division at which they occurred. First-level communities (1 and 2) and second-level communities (1a, 1b, 2a and 2b) are numbered and 'N' is the number of sample sites associated with each division.	122
Figure 4.14 The distribution of sample sites corresponding to the TWINSpan-defined communities: a) 1 and 2; and b) 1a, 1b, 2a and 2b; across the wetland and reedbed habitats identified from the June 2012, 2013 and July 2014 survey data.	124
Figure 4.15 Number of TWINSpan-defined wetland and reedbed communities at each of the three elevational ranges used in ordination: a) First-level communities 1. <i>Fen Meadow</i> and 2. <i>Peat Mire</i> ; and Second-level communities b) 1a. <i>Groundwater Mire</i> , 1b. <i>Sedge Mire</i> , 2a. <i>Rush Pasture</i> and 2b. <i>Rush Mire</i>	125
Figure 4.16 Distribution of TWINSpan-defined second-level wetland and reedbed communities 1a, 1b, 2a and 2b in relation to regularly and infrequently grazed areas and predator control zones.	126
Figure 4.17 Dendrogram TWINSpan result for lake plant species data. Plant species in boxes represent indicator species for the point of division at which they occurred. Communities are numbered and 'N' is the number of sample sites associated with each division. First-level communities (1 and 2) and second-level communities (1a, 1b, 2a and 2b) are numbered and 'N' is the number of sample sites associated with each division.	127
Figure 4.18 GIS showing the distribution of sample sites corresponding to the TWINSpan-defined a) Communities 1 (Littoral Coastal), and 2 (Benthic Coastal), and b) Communities 1a (Benthic Lowland), 1b (Littoral Lowland), 2a (Shallow Coastal)	

	and 2b (Shallow Lowland) across the open water habitat of the lake identified from the June 2012 and 2013 survey data.	129
Figure 4.19	Number of TWINSPAN-defined lake communities at each of the three depth classes used in ordination: a) First-level communities 1. <i>Littoral Coastal</i> and 2. <i>Benthic Coastal</i> ; and Second-level communities b) 1a. <i>Benthic Lowland</i> , 1b. <i>Littoral Lowland</i> , 2a. <i>Shallow Coastal</i> and 2b. <i>Shallow Lowland</i>	129
Figure 4.20	a) pH and b) conductivity ($\mu\text{S cm}^{-1}$) measured in-situ in June 2013 at the hydrological monitoring sites located across the lake, wetland and dune systems.	131
Figure 4.21	Nitrate, total phosphate and soluble reactive phosphorus (mg L^{-1}) measured in-situ in June 2013 at the hydrological monitoring sites located across the lake, wetland and dune systems.	132
Figure 4.22	Annual total precipitation (mm) and mean annual temperature ($^{\circ}\text{C}$) from 1957 to 2014 recorded at the Malin Head meteorological station (ECAD, 2015).	134
Figure 4.23	Daily total precipitation (mm), mean daily air temperature ($^{\circ}\text{C}$) and daily total evapotranspiration (mm) over the 2012 to 2014 survey period recorded at the Sheskinmore Lough weather station.	136
Figure 4.24	a) Daily average wind speed ms^{-1} over the 2012 to 2014 survey period recorded at the Sheskinmore Lough weather station. Wind rose plots showing the b) mean wind speed (m/s) and c) highest wind speeds (m/s) based on the Beaufort Scale with wind direction recorded during the 10-minute period at each 30-minute recording interval during the June 2012 to June 2014 survey period.....	138
Figure 4.25	Mean monthly a) absolute precipitation (mm), b) temperature ($^{\circ}\text{C}$), c) absolute evapotranspiration (mm), and d) wind speed (ms^{-1}) over the 2012 to 2014 survey period recorded at the Sheskinmore Lough weather station.	139
Figure 4.26	Digital elevation models (DEMs) of the study site: a) 1m resolution DEM of the study area based on contour data with habitats labelled and the three key habitat areas outlined (wetland, reedbed and open water area of the lake), and b) 1m resolution DEM of the lake and wetland based on dGPS topographic and bathymetric survey data.	140
Figure 4.27	Elevation ranges in metres above sea level (mOD) of the habitats at Sheskinmore Lough SPA based on contour data. Note that maximum and minimum elevation values are included within the graph, as is the mean, which is represented by the line inside each bar. Also note the 'Open Water' habitat represents the full range of the Sheskinmore Lough and Sandfield Lough elevation ranges that are also included as separate habitat areas.	141
Figure 4.28	a) The location of the on-site weather station (WSt) and the hydrological monitoring sites contributing data to the two-year survey (18 th June 2012 to 17 th June 2014). The sites include those in the inflows (Duvoe L1; Abberachrin L2), Sheskinmore Lough (L3), the sluice outflow (L4), Sandfield Lough (L6), the wetland system (W1-3; W5-6), the surrounding area (S3), the dunes (D1) and the beach pond system (D2). Note that colourless sites L5, W4, S1, S2, S4 and S5 are not included in this survey. b) Elevational distribution of the hydrological monitoring sites from D2 across the lake and wetland system from west to east for the two-year survey period.	142
Figure 4.29	The elevations of the sluice gate when it is open and closed relative to the elevation of hydrological monitoring site L4 in the outflow. Note when the sluice is open water passes underneath the gate, an event that occurs below the water level and is therefore not visible in this photo.	143

- Figure 4.30 Daily total precipitation (mm) measured at the on-site weather station and daily mean water levels (mm) relative to Ordnance Datum (OD) at Malin Head, in the a) lake system (L1-4; L) and b) in the wetland system (S3; W1-3; W5-7; D1-2) recorded during the two-year monitoring period from 18th June 2012 to 17th June 2014.144
- Figure 4.31 Range in mean daily water levels (mm) relative to Ordnance Datum at each site across a) the lake system at L1-4 and L6 and b) the wetland system at S3, W1-3, W5-7 and D1-2, during the two-year monitoring period from 18th June 2012 to 17th June 2014. Note the overall mean water level (mmOD) for the survey period is labelled for each site. Note differences in y-axis scales.145
- Figure 4.32 Daily total precipitation (mm) measured at the on-site weather station and daily mean water levels relative to Ordnance Datum (OD) at Malin Head (mm) in the a) hydrologically managed Sheskinmore Lough (L3) and b) unmanaged Sandfield Lough (L6) during the two-year monitoring period from 18th June 2012 to 17th June 2014. Note: the red arrows indicate where the sluice was opened during the survey period.146
- Figure 4.33 Daily mean water levels relative to the ground surface (mm), measured in the lake system at L1-4 and L6 during the two-year monitoring period from 18th June 2012 to 17th June 2014.147
- Figure 4.34 Daily mean water levels relative to the ground surface, measured during the two-year monitoring period from 18th June 2012 to 17th June 2014 in the dipwells installed across the wetland system (S3; W1-3; W5-7; D1), and in the hydrological monitoring site in the dune pond (D2). Note differences in y-axis scales.148
- Figure 4.35 Hourly total precipitation (mm) measured at the on-site weather station and hourly mean water levels (mm) relative to Ordnance Datum (OD) at Malin Head, in the a) lake system (L1-4; L6) and b) in the wetland system (S1-3; W1-7; D1-2) recorded during the six-week monitoring period from 7th September 2012 to 22nd October 2012.150
- Figure 4.36 Range in hourly mean water levels (mm) relative to Ordnance Datum at each site across a) the lake system at L1-4 and L6 and b) the wetland system at S1-3, W1-7 and D1-2, during the six-week monitoring period from 7th September to 22nd October 2012. Note the overall mean water level (mmOD) for the survey period is labelled for each site and also note the differences in y-axis scales.151
- Figure 4.37 Hourly total precipitation (mm) measured at the on-site weather station and hourly mean water levels relative to the ground surface (mm), measured in the lake system at L1-4 and L6, across and around the wetland and dune system at S1-3, W1-7 and D1-2 during the six-week monitoring period from 7th September to 22nd October 2012. Note differences in y-axis scales.152
- Figure 4.38 Hourly total precipitation (mm) measured at the on-site weather station and hourly mean water levels (mm) relative to Ordnance Datum (OD) at Malin Head, in the a) lake system (L1-4; L6) and b) in the wetland system (S1-3; W1-3; W5-7; D1-2) recorded during the four-week monitoring period from 1st to 31st July 2013.154
- Figure 4.39 Range in hourly mean water levels (mm) relative to Ordnance Datum at each site across a) the lake system at L1-4 and L6 and b) the wetland system at S1-3, W1-3, W5-7 and D1-2, during the four-week monitoring period from 1st July to 31st July 2013. Note the mean daily average water level (mmOD) is labelled for each site and also note the differences in y-axis scales.154
- Figure 4.40 Hourly total precipitation (mm) measured at the on-site weather station and hourly mean water levels relative to the ground surface (mm), measured in the

- lake system at L1-4 and L6 and in the dipwells installed across the wetland system (S1-3; W1-3; W5-7; D1-2) during the four-week monitoring period from 1st to 31st July 2013. Note differences in y-axis scales.155
- Figure 4.41 a) The location of the on-site weather station (WSt) and the hydrological monitoring sites contributing data to the seven-day survey (18th to 24th June 2012). The sites include those in the inflows (Duvoge L1; Abberachrin L2), Sheskinmore Lough (L3), the sluice outflow (L4), the downstream estuary section of the outflow (L5), Sandfield Lough (L6), the wetland system (W1-7), the dune ponds (S1-2), the surrounding wet grassland (S3-5), the dune blowout (D1) and the beach pond system (D2). b) Elevational distribution of the hydrological monitoring sites from D2 across the lake and wetland system from west to east for the seven-day survey period.156
- Figure 4.42 Quarter-hourly total precipitation (mm) measured at the on-site weather station and quarter-hourly mean water levels (mm) relative to Ordnance Datum (OD) at Malin Head, in the a) lake system (L1-6) and b) in the wetland system (S1-5; W1-7; D1-2) recorded during the seven-day monitoring period from 18th to 24th June 2012.157
- Figure 4.43 Range in quarter-hourly mean water levels (mm) relative to Ordnance Datum at each site across a) the lake system at L1-6 and b) the wetland system at S1-5, W1-7 and D1-2, during the seven-day monitoring period from 18th to 24th June 2012. Note the mean daily average water level (mmOD) is labelled for each site and also note the differences in y-axis scales.158
- Figure 4.44 Quarter-hourly mean water levels relative to the ground surface, measured during the seven-day monitoring period (18th to 24th June 2012) in the lakes (L3, L6), inflows (L1-2) and outflow (L4-5), dune ponds (S1-2), surrounding wet grassland (S3-5), wetland (W1-7), dune blowout (D1), and beach pond (D2). Note differences in y-axis scales.159
- Figure 4.45 a) PCA showing the relationships between the quarter-hourly water levels, and b) linear regression (r^2) values showing the significance of the quarter-hourly water level relationships at the hydrological monitoring sites across the lake (L1-6), wetland (W1-7), dune system (D1-2) and surroundings (S1-5) observed during the seven-day monitoring period from 18th to 24th June 2012. PCA eigenvalues: Axis 1 = 64.3%; Axis 2 = 14.8%.160
- Figure 4.46 Distribution of the WETMEC hydrological groups and subgroups across the wetland and dune systems in relation to the wetland, dune and surrounding hydrological monitoring sites. WETMEC 3: Buoyant Weakly Minerotrophic Surfaces ('Transition Bogs'); WETMEC 6f: Surface Water Percolation Water Fringe; WETMEC 9: Groundwater-Fed Bottoms; WETMEC 12: Fluctuating Seepage Basins; WETMEC 20b: Percolation Water Fringe.167
- Figure 4.47 Distribution of lake flooding frequency ranges, calculated as the percentage of the two-year survey period (18th June 2012 to 17th June 2014) inundated by more than 10cm of water, relative to the open water habitat of the lake, and the wetland and reedbed systems.170
- Figure 4.48 PCA showing the relationships between environmental variables measured in June 2013 across the study site and the TWINSPAN-defined a) first-level and b) second-level communities of the wetland and reedbed, and c) PCA statistics.....173
- Figure 4.49 SEV soil waterlogging (dark blue shaded area) and SEV soil drying (light blue shaded area) calculated from 18th June 2012 to 17th June 2013 at the hydrological monitoring sites in the wetland: W1-3; and W5-7. The horizontal lines represent threshold depths for the particular soil type (-48mm to -55mm for peat; -180mm

to -320mm for sandy peat): soil waterlogging threshold (upper line); soil drying threshold (lower line). Note the absence of soil waterlogging from October to February as it is only cumulated during the period of active vegetation growth (March to September), when plants are most sensitive to the oxygen status in their root zone (Gowling et al, 2002). Note also the varying vertical scales.....	180
Figure 4.50 Water regime for all of the 6 wetland hydrological monitoring sites as defined by their SEV waterlogging and SEV soil drying calculated as mm per day, recorded from 18th June 2012 to 17th June 2013.	181
Figure 4.51 Water regime of each TWINSPAN-defined community a) first-level communities and b) second-level communities, based on interpolation between the 120 wetland and reedbed vegetation survey sites and the 6 wetland hydrological monitoring sites (W1-3 and W5-7) as defined by their SEV waterlogging and SEV soil drying calculated as mm per day, recorded from 18th June 2012 to 17th July 2016.	182
Figure 4.52 Schematic of submerged macrophyte species identified on 14th July 2014 along the four transects across the a) northern and b) southern channel feature, and at the c) southern and d) north-eastern margins within the open water habitat of the lake, in relation to the depth of the minimum and average monthly water levels relative to the maximum monthly water levels defined using the system designed by Cui et al (2010).	184
Figure 4.53 Maximum water depth range observed across the entire Sheskinmore Lough open water area during the survey period relative to the preferred water depths of each of the submerged macrophyte species identified in June 2012 and July 2013. Note the dark grey shaded area which has been included to aid comparison.	185
Figure 4.54 Distribution of exposure frequencies, calculated as the percentage of the survey period (18th June 2012 to 17th June 2013) inundated by less than 10cm of water, relative to the four transects (within the A) northern and B) southern channel feature, and on the C) southern and D) north-eastern margins of the open water habitat of the lake) and the wetland and reedbed systems.	187
Figure 4.55 Conceptual ecohydrological model highlighting the key relationships between the ecology and hydrology of the wetland and lake system. The accompanying map shows the distribution of the system zones in the model, which include the terminal wetland, transition wetland (which includes the reedbed) and lake.	189
Figure 5.1 Locations of the exploratory sediment cores retrieved in June 2012 from hydrological monitoring sites across the wetland (S3, W1, W2, W3, W4, W6, W7) and within Sheskinmore (L3, L3.2, L3.6) and Sandfield (L6) Loughs. Note W1 and W4 did not undergo further analyses, only sediment description, due to contamination.	191
Figure 5.2 Stratigraphy of the exploratory sediment cores extracted from sites across the lake and wetland system relative to a) the ground surface and b) OD. Cores are ordered west to east (from left to right) with W1 being the most westerly site and L6 being the most easterly site. Note Appendix 1.4 provides more detailed measurements of the sections.	193
Figure 5.3 Exploratory core grain size distribution, mean grain size, sorting, percent organic and percent carbonate content in relation to depth below the ground surface. Note, data are presented vertically to aid visualisation; samples within the individual cores are vertically related, but there is no vertical association between cores.	195

Figure 5.4 Macrofossil abundance identified within the top 30 cm of sediment taken from a) Sheskinmore Lough (L3) and b) Sandfield Lough (L6).	196
Figure 5.5 Diatom species abundance identified within the top 30cm of sediment taken from a) Sheskinmore Lough at L3 and b) Sandfield Lough at L6.....	197
Figure 5.6 Sediment stratigraphies of the two Sheskinmore Lough lake cores with elevation. Images show the split core surface.	200
Figure 5.7 Lake Core 1 grain size distribution, mean and sorting relative to elevation (mOD) and depth below the lakebed (cm).	201
Figure 5.8 Lake Core 2 grain size distribution, mean and sorting relative to elevation (mOD) and depth below the lakebed (cm).	202
Figure 5.9 The percent organic and percent carbonate content in Core 1 and Core 2 in relation to depth (m) below the lake bed. Note the different y-axis scales.	203
Figure 5.10 Venn diagram showing the 59 aquatic, marginal and terrestrial plant species and animal species/class/order of macrofossils that were identified only in Core 1 (27), only in Core 2 (9), and in both Core 1 and Core 2 (23).	205
Figure 5.11 Pie charts showing the proportions of the 10 most abundant macrofossils identified in each of the two lake cores extracted in June 2013.	205
Figure 5.12 Downcore variations in relative abundance of individual macrofossils, and total macrofossil abundance and richness in relation to TWINSpan defined zones identified in Cores 1 and 2.	207
Figure 5.13 Venn diagram showing the 97 freshwater and freshwater-brackish lake core diatom species identified: only in Core 1 (38); only in Core 2 (3); and in both Core 1 and Core 2 (56). The diagram also indicates whether the primary habitat niche association is pelagic, periphytic, epilithic or benthic.	208
Figure 5.14 Pie charts showing the proportions of the 10 most abundant diatom species identified in the two lake cores extracted in June 2013.	209
Figure 5.15 Downcore variations in diatom percentage of selected taxa (> 2% abundance in at least three samples), and total diatom abundance and species richness in relation to TWINSpan defined diatom communities in Cores 1 and 2. Note Appendix 1.7 details the relative abundance of all diatom taxa identified in the two cores.	211
Figure 5.16 Summary of downcore profiles in Core 1 and Core 2 including sediment characteristics and key diatom and macrofossils results.	214
Figure 5.17 Summary of downcore profiles in Core 1 with CalPal calibrated ¹⁴ C radiocarbon dates overlain.	217
Figure 5.18 The 1775 Murdoch Mackenzie map of the Inishkeel Parish and the 1777 George Taylor & Andrew Skinner map, the left panel of which marks the main road north from Killybegs to Inishkeel via Ardara (MacKenzie, 1775; Taylor & Skinner, 1778). Note that no lake is recorded on either map in the vicinity of Sheskinmore Lough, so the approximate position has been marked in blue.	219
Figure 5.19 The 1801 James McParlan Statistical Survey of The County Of Donegal map (McParlan, 1802). Note that no lake is recorded on the map in the vicinity of Sheskinmore Lough, so the approximate position has been marked in blue.	220
Figure 5.20 Section of sheet V, Farland Point to Donegal Bay, from the 1835 UKHO admiralty chart of Ireland surveyed by Captain Mudge, and a section of the 1:25,000 first edition of the '1 inch to 1 mile' OS map from 1860, both held in the British Library (Burningham, 1999). Sheskinmore Lough is outlined in blue.	221

Figure 5.21 Section of the second edition of the '1 inch to 1 mile' OS map at 1:25,000 scale from 1901, and a section of the second edition OS 1:10,560 map published in 1907 but based on revisions conducted in 1905 (Burningham, 1999). Sheskinmore Lough is outlined in blue.	222
Figure 5.22 Sections of aerial photographs from 1951 (Irish Air Corps), 1977 (Ordnance Survey of Ireland) and 1995 (Ordnance Survey of Ireland) (Burningham, 1999; Ordnance Survey Ireland, 2016). Sheskinmore Lough is outlined in blue. Note the exposure and contrast of the Sheskinmore Lough in the 1951 photograph has been edited to highlight the lake bed, shown in the inset.	223
Figure 5.23 Satellite images 1:10,000 from 2000, 2005 (both from Ordnance Survey Ireland) and 2012 (from Bing) showing Sheskinmore Lough at the centre of each image (Bing, 2016; Ordnance Survey Ireland, 2016).	224
Figure 5.24 Stages in the development of Sheskinmore Lough from the 1500s to present.	226
Figure 5.25 Timeline summarising the historical phases identified from the stratigraphic record in relation to key climate events, sunspot cycles and Irish famines.	227
Figure 6.1 Hydrological processes simulated by MIKE SHE (DHI, 2005; p1).	239
Figure 6.2 The numerical engines for each hydrological process available in MIKE SHE (Graham & Butts, 2005; p3).	239
Figure 6.3 The <i>whole</i> and <i>immediate</i> catchments at Sheskinmore Lough.....	242
Figure 6.4 Comparison of monthly total precipitation (mm month ⁻¹) data recorded at the Sheskinmore Lough and regional Malin Head, Finner and Ballyhaise weather stations (and their respective locations in Ireland) during the first year of the hydrological monitoring period (Met Éireann, 2015).	243
Figure 6.5 Processed topographic map (elevation mOD) of the a) <i>whole</i> model catchment (grid size 70m ²) and b) <i>immediate</i> model catchment (grid size 40m ²).	245
Figure 6.6 Hypsometric curves for the Sheskinmore topography in the a) <i>whole</i> and b) <i>immediate</i> catchments.	245
Figure 6.7 Processed vegetation areas in the a) <i>whole</i> model catchment (grid size 70m ²) and b) <i>immediate</i> model catchment (grid size 40m ²).	246
Figure 6.8 MIKE 11 models of the a) <i>whole</i> and b) <i>immediate</i> catchments.	248
Figure 6.9 Cross sections at the sub-branch connections of the a) Abberachrin, b) Duvoige and c) Outflow rivers.	249
Figure 6.10 Longitudinal sections of the a) Abberachrin and Outflow and b) Duvoige and Outflow rivers labelled with sub-branch chainages and the relative position of the lake.	250
Figure 6.11 Observed time series of sluice operation (opening and closing) during the hydrological monitoring period from 18 th June 2012 to 17 th June 2014.	250
Figure 6.12 Processed Unsaturated Zone soil types in the a) <i>whole</i> model catchment (grid size 70m ²) and b) <i>immediate</i> model catchment (grid size 40m ²).	252
Figure 6.13 Processed Saturated Zone sand layer in the a) <i>whole</i> model catchment (grid size 70m ²) and b) <i>immediate</i> model catchment (grid size 40m ²).	253
Figure 6.14 Hypsometric curves for the Sheskinmore saturated zone sand layer depth in the a) <i>whole</i> and b) <i>immediate</i> catchments.	253
Figure 6.15 Comparison of daily and monthly observed and simulated discharge in the	

Abberachrin (a) and c)) and Duvoge (b) and d)) rivers in the <i>whole</i> model for the period of 18 th June 2012 to 17 th June 2013. Note the model stabilisation period from 18 th June to 17 th September 2012.	256
Figure 6.16 Comparison of a) daily and b) monthly observed and simulated lake water level in the <i>immediate</i> model for the period of 18 th June 2012 to 17 th June 2013. Note the model stabilisation period from 18 th June to 17 th September 2012.	258
Figure 6.17 Comparison of daily observed and simulated daily discharge in the a) Abberachrin and b) Duvoge rivers in the <i>whole</i> model for the period of 18 th June 2013 to 17 th June 2014. Note the model stabilisation period from 18 th June to 17 th September 2013.	259
Figure 6.18 Comparison of a) daily and b) monthly observed and simulated lake water level in the <i>immediate</i> model for the period of 18 th June 2013 to 17 th June 2014. Note the model stabilisation period from 18 th June to 17 th September 2013.	260
Figure 6.19 Coverage of UKCIP 2009 probabilistic projections for future climate in the river basin region of northwest Ireland.	263
Figure 6.20 Monthly mean absolute catchment precipitation and evapotranspiration for the baseline scenario at probabilities between 10% and 90% for each emissions scenario (L = Low, M = Medium, H = High) for the 2050s and 2080s.	265
Figure 6.21 Projected mean absolute annual, summer and winter catchment precipitation and evapotranspiration for the baseline scenario at probabilities between 10% and 90% for each emissions scenario (Low, Medium, High) for the 2050s and 2080s. Note the different y-axis scales.	266
Figure 6.22 Projected mean daily water level in Sheskinmore Lough for the baseline scenario at probabilities between 10% and 90% for each emissions scenario (L = Low, M = Medium, H = High) for the 2050s and 2080s.	268
Figure 6.23 Projected mean monthly water level in Sheskinmore Lough for the baseline scenario at probabilities between 10% and 90% for each emissions scenario (L = Low, M = Medium, H = High) for the 2050s and 2080s.	269
Figure 6.24 Projected mean annual, summer and winter water level in Sheskinmore Lough for the baseline scenario at probabilities between 10% and 90% for each emissions scenario (Low, Medium, High) for the 2050s and 2080s. Note the different y-axis scales.	269
Figure 6.25 Impacts of future climate change on overland water depth (m) across the Sheskinmore Lough lake and wetland system for a range of scenarios based on the maximum (Max), minimum (Min) and mean (Mean) modelled lake water level under each emissions scenario (Low, Medium and High) for the 2050s and 2080s. Note the lake open water and reedbed habitats are outlined in blue and green, respectively, while the remainder of the modelled area encompasses the wetland habitat.	271
Figure 6.26 Projected mean daily water level in Sheskinmore Lough for the <i>as is</i> , <i>fully open</i> and <i>fully closed</i> scenarios under the baseline (<i>as is</i>) climate conditions.	273
Figure 6.27 Impacts of the <i>as is</i> (As Is), <i>fully open</i> (Open) and <i>fully closed</i> (Closed) management scenarios on maximum (Max), minimum (Min) and mean (Mean) overland water depth across the Sheskinmore Lough lake and wetland system under current climate conditions. Note the lake open water and reedbed habitats are outlined in blue and green, respectively, while the remainder of the modelled area encompasses the wetland habitat.	273
Figure 6.28 Projected mean monthly water level in Sheskinmore Lough under the three	

initial management scenarios (*as is*, *fully open*, *fully closed*) in the current climate, and also under the future climate change emissions scenarios (L = Low, M = Medium, H = High) at probabilities between 10% and 90% for the 2050s and 2080s.275

Figure 6.29 Impacts of the combined effect of future climate change and hydrological management on overland water depth across the Sheskinmore Lough lake and wetland system for a range of scenarios based on the maximum (Max), minimum (Min) and mean (Mean) modelled lake water levels for each management scenario (*as is*, *fully open* and *fully closed*) under the Medium emissions scenario for the 2050s and 2080s. Note the lake open water and reedbed habitats are outlined in blue and green, respectively, while the remainder of the modelled area encompasses the wetland habitat.277

Figure 6.30 Grime's CSR triangle and climate change impacts on shoreline habitats with fertility and disturbance axes (Adapted from Abraham, 2008; p35).280

Figure 6.31 Grime's CSR triangle, climate change impacts on shoreline habitats with fertility and disturbance axes, and potential management options to allow adaption to climate change (Adapted from Abraham, 2008; p37).286

List of tables

Table 1.1 Simple wetland classification system based on ecohydrological and locational characteristics (adapted from Keddy, 2010; p18).	33
Table 1.2 Ramsar Convention classification system of wetland type (Ramsar, 2009).	36
Table 1.3 Comparisons of the estimated extent of global wetlands by climatic zone (adapted from Mitsch & Gosselink, 2007; p46).....	37
Table 1.4 Lake type based on formative origins defining the Hutchinson (1957) classification system (adapted from Thomas et al, 1996; 326).	43
Table 1.5 Lake classification based on morphology (adapted from Löffler, 2004; p34).	44
Table 1.6 Characteristics of shallow and deep lakes (adapted from Cooke et al, 2001; p43).	44
Table 1.7 Lake classification based on geochemistry (adapted from Löffler, 2004; p36-46).	45
Table 1.8 The average rates of lake macrophyte primary production. Note the seasonal maximum biomass is approximate and the rates of production vary greatly (adapted from Wetzel, 1999: 279).	56
Table 2.1 The largest lakes in Ireland, their location and dimensions.....	61
Table 2.2 The five main morphological coastal lagoon types in Ireland, their location, the number surveyed and their dominance in terms of the coastal lake habitat surveyed (Adapted from Oliver, 2007; p34).	64
Table 2.3 Coastal lagoons identified in northwest Ireland during the NPWS survey, their size and type (Adapted from Oliver, 2007).	65
Table 2.4 The typical species found in coastal lakes according to conductivity (Adapted from Oliver, 2007: p24).	66
Table 2.5 The relative proportions of different habitat areas within Sheskinmore Lough SPA (Adapted from NPWS, 2005: p1).	72
Table 2.6 The relative proportions and intensities of the various positive and negative anthropogenic impacts on flora and fauna identified within the Sheskinmore Lough SPA (NPWS, 2005).	74
Table 3.1 The full suite of meteorological metrics available with the Davis VantagePro2 automatic weather station (AWS).	87
Table 3.2 Gaps in meteorological data observed during the second year of the monitoring period (18 th June 2013 to 17 th June 2014).	88
Table 3.3 Discharge and stage in the Duvoqe inflow (L1) and the Abberachrin inflow (L2) following constant rate salt injection in June 2013.....	93
Table 3.4 Discharge and stage in the outflow (L4) following constant rate salt injection in June 2013.	95
Table 3.5 Specification of samples submitted to the NERC Radiocarbon Facility for initial radiocarbon dating (A17 and A67) and the subsequent pair of samples (A25 and A63) submitted to the NERC Radiocarbon Facility for radiocarbon dating.	104
Table 4.1 Wetland and reedbed plant communities identified during the June 2012, 2013 and July 2014 surveys assessed through TWINSPAN, their associated species and National Vegetation Classification (NVC) and Natura 2000 'EUR27' codes (Elkington et al, 2001; EC, 2007). See Appendix A. for full descriptions.	122

Table 4.2 Lake plant communities identified during the June 2012 and 2013 surveys assessed through TWINSPAN, their associated species and National Vegetation Classification (NVC) and Natura 2000 'EUR27' codes (Elkington et al, 2001; EC, 2007). See 1.2 for full community descriptions.	127
Table 4.3 Summary of the key daily and monthly weather characteristics over the 2012 to 2014 survey period recorded at the Sheskinmore Lough weather station.	137
Table 4.4 The dates when the sluice was opened and closed during the 730-day hydrological monitoring period from 18 th June 2012 to 17 th June 2014 (Emer Magee pers. comm. 8 th August 2014).	143
Table 4.5 Hydromorphological classification of fen wetlands as defined by Goode, 1977 (Adapted from Lloyd et al, 1993). Note the red box indicates the hydromorphological classification of the wetland system at Sheskinmore Lough.	163
Table 4.6 Wetland situation types and component hydrotopographical elements for wetland classification (adapted from Wheeler & Shaw, 1995). +++ particularly characteristic of the situation type; ++ sometimes occurs within the situation type; + of minor importance, or peripheral.	164
Table 4.7 Hydrogeological classification of fen wetlands as defined by Lloyd et al, 1993 (Adapted from Wheeler et al, 2004). Note the red boxes indicate the hydrogeological and hydromorphological classification of the wetland at Sheskinmore Lough.	165
Table 4.8 WETMEC hydrological classification groups and subgroups associated with each hydrological monitoring site across the central and surrounding wetland (W1-7 and S1-5) and dune (D1-2) systems (Wheeler et al, 2009; Kimberley & Coxon, 2013).	166
Table 4.9 Descriptions of the WETMEC hydrological classification groups and subgroups associated with the hydrological monitoring sites across the central and surrounding wetland (W1-7 and S1-5) and dune (D1-2) systems. See Appendix C. for full descriptions (Wheeler et al, 2009; Kimberley & Coxon, 2013).	166
Table 4.10 Classification of the lake at Sheskinmore Lough using the tiered typology proposed by Northern Ireland and also applied by Ireland to explain biological variation in lakes (Adapted from UKTAG, 2003 and Free et al, 2005).	169
Table 4.11 Hydrological classification of lake water levels based on monthly average and above-optimum/sub-optimum water levels observed during the two-year monitoring period from 18 th June 2012 to 17 th June 2014 (Cui et al, 2010). Note Class A = >140mm average monthly water level; Class B = 120 -140mm average monthly water level; Class C = <120mm average monthly water level; Class a = Over double the no. of above-optimum days compared to sub-optimum days; Class b = Less than double the no. of above-/sub-optimum days compared to sub-/above-optimum days; Class c = Over double the no. of sub-optimum days compared to above-optimum days.	169
Table 4.12 The hydrochemical environment of Sheskinmore Lough and Sandfield Lough (recorded in June 2013) in the context of Irish and European lake hydrochemical ranges observed from three studies that have surveyed the environment of 774 Irish lakes (200 lakes in the Republic of Ireland (Aherne et al, 2002) 574 lakes in Northern Ireland (Heegaard et al, 2001)), and 86 lakes in Europe (Nöges et al, 2003).	175
Table 5.1 Radiocarbon (¹⁴ C) dating results for Core 1 [calibration performed by CalPal ¹ and Calib 7.1 ²].	215

Table 5.2 Description of the lithostratigraphy and paleoecology in the Shaw & Cater (1994) core extracted from the wetland to the northeast of Sheskinmore Lough.	225
Table 6.1 Summary of the MIKE SHE processes, data and methods relevant to the present study (Adapted from DHI, 2005).	240
Table 6.2 Key areas of MIKE SHE application and their references (Adapted from Graham & Butts, 2005; and Singh, 2010).	241
Table 6.3 Comparison of the original and processed land use vegetation type data, and their respective Leaf Area Index and root depth values for both the <i>whole</i> and <i>immediate</i> catchment models.	246
Table 6.4 Details of the river network features for the MIKE 11 HD <i>whole</i> and <i>immediate</i> catchment models.	247
Table 6.5 Control structure (sluice) configuration details in the <i>immediate</i> catchment MIKE 11 model.	251
Table 6.6 Comparison of the original and processed Unsaturated Zone soil areas for both the <i>whole</i> and <i>immediate</i> models.	252
Table 6.7 Hydraulic parameters and the initial values specified for the Unsaturated Zone soil types.	252
Table 6.8 Hydraulic parameters specified for the Saturated Zone soil types.	253
Table 6.9 Initial and final calibrated parameter values in the MIKE SHE and MIKE 11 models.	255
Table 6.10 Henriksen performance indicators for post-stabilisation model calibration performance (Henriksen et al, 2008).	257
Table 6.11 Henriksen performance indicators for post-stabilisation model validation performance (Henriksen et al, 2008).	260
Table 6.12 Range of projected mean daily and monthly lake water levels under each <i>initial</i> (current climate) scenario and also under each emissions scenario (L = Low, M = Medium, H = High) across the range of management scenarios (<i>as is</i> , <i>fully open</i> and <i>fully closed</i>), at probabilities between 10% and 90% for the 2050s and 2080s....	276
Table 6.13 Summary of the likely future ecohydrological impacts per season and sluice status from the combined influence of climate change and hydrological management at Sheskinmore Lough.	288
Table 7.1 Suggested areas of development for further research on species, communities and environment of selected habitats at Sheskinmore Lough SPA.	300

Acknowledgements

I am particularly indebted to Dr Helene Burningham, my principal supervisor, not only for giving me the opportunity to undertake the PhD that is totally 'me', but also for her unrivalled passion, knowledge, dedication and encouragement from start to finish. Without her creativity when the natural world presented some unexpected challenges and her always spot on advice this PhD would not have been possible. I am also extremely grateful to Dr Julian Thompson, my secondary supervisor, for his invaluable help with the intricacies of MIKE SHE and for his patient advice when the model 'got the better of me'!

This PhD would also not have been possible without the support of colleagues in Ireland. In particular, I would like to express sincere thanks to the National Parks and Wildlife Service (NPWS), especially to Tim Roderick, Emer Magee and Andrew Speer, who not only kindly granted me permission to carry out this project, but provided an constant font of knowledge. Additional thanks must go to Emer for her dedicated support with the weather station and ongoing logistics, and special thanks must go to Andrew for his many trips down south to accompany me on my many over- and under-water expeditions! I would also like to thank all the landowners who gave me access, especially: Seamus Meehan, Michael Kennedy, Aidan Campbell, and BirdWatch Ireland. Lastly, I would like to thank, Malcolm McClure for sharing some truly fascinating geological and archaeological insights.

I would like to extend my sincere thanks to Dr Vivienne Jones, for her invaluable guidance when writing my radiocarbon dating application. Thanks to her, I am also able to thank Dr Luz Maria Cisneros-Dozal and Dr Steven Moreton of the NERC Radiocarbon Facility at East Kilbride for their help and advice. Similarly, I must also thank Elizabeth Paull and Alison Callaway at Aquatec Group, for their logistical support during my Aquatec award tenancy.

Given the multifaceted nature of this PhD, I am truly grateful to the many people who have helped me at various points along the way, including: Dr Carl Sayer, for his unmatched enthusiasm and expert eye with a microscope; Roger Flower and Ewan Shilland for their specialist identification skills; David Maher-McWilliams, Katherine Bolles and Stephanie Goodrich for their elbow grease during fieldwork; Dr Handong Yang, Ian Patmore, Janet Hope, Kevin Roe and Tula Maxted, for their assistance in the laboratory; and my fellow PhD students Amanda Green, Arnaud Duranel and Helen Greaves for going out of their way to help at various points. Thanks must also go to all the other members of staff in the UCL Department of Geography who provided vital logistical support, namely: Ajay Chauhan, Linda Fuller, Maria Rodriguez, Miles Irving, Susan Hennessy and Dr Suse Keay.

Last but by no means least, I would like to extend a huge thanks to my family and friends who have been my rock throughout this PhD. To my mother and fieldwork buddy, Barbara Gardner, who was always there to keep me on the straight and narrow and lift my spirits when things (quite often!) seemed impossible. To my (two) fathers, James Gardner and Martin Critchell, for all the things they have done, and will continue to do, for me. And finally, to my husband-to-be, Felix Meston, for his unrelenting love and support, for putting up with me when times got tough, and for always believing in me. Words cannot express how grateful I am.

1 Introduction

1.1 Aquatic environments

Freshwater aquatic environments are essential for life, but they also constitute a valuable natural resource in economic, cultural, aesthetic, scientific and educational terms. In the late 20th century, a growing appreciation of servicing and supporting functions for wildlife and humans that these ecosystems provide has led to acknowledgment that their loss and degradation is a major cause for concern (Williams, 1990; Thompson & Hollis, 1997; Mitsch & Gosselink, 2000; Finlayson & Moser, 1992; Mitsch et al, 1994; CEC, 1995). Freshwater systems are experiencing declines in biodiversity far greater than those in most terrestrial ecosystems. If trends in demand for water remain unaltered, and species losses continue at current rates, the opportunity to conserve much of the remaining freshwater biodiversity will vanish. Freshwater systems are sensitive environments, and impacts from climate change and anthropogenic pressures are likely to have important consequences for their conservation (Kundzewicz et al, 2007; Bates et al, 2008; Thompson 2012).

Lowland, shallow, lake systems comprise a complex suite of wetland and open-water habitats and species that are a product of their geomorphic evolution, in combination with hydrological and terrestrial forcing. These systems usually comprise a wealth of different vegetation communities that reflect subtle changes in elevation, proximity to the water table, hydroperiod, microclimates, and connection to terrestrial hinterland. They often have a long history of use by humans as they provide easy access to freshwater and are associated with high vegetative productivity. The consequence of this is that many lowland aquatic environments have suffered significant degradation through pollution (mostly eutrophication) and water management (introduction of artificial hydroregimes intended to provide more consistent or managed water supply).

Although water management has played a significant part in the degradation of aquatic environments, hydrological management initiatives are an important component of conservation measures to deliver sustainable habitat restoration and rehabilitation. Understanding the impacts of these schemes, and improving the ability to predict impacts on these systems in the future, are therefore required in order to develop management schemes that will achieve their goals, avoid undesirable outcomes and effectively target the often limited resources available to conservation practitioners (Brooks et al, 1991; Refsgaard et al, 1992; Jain et al, 1992; Thompson & Hollis, 1995; Lorup et al, 1998; Al-Khudhary et al, 1999; Karvonen et al, 1999; Christiaens & Feyen, 2001; Thompson et al, 2004; McMichael et al, 2006).

A sound scientific basis is crucial, particularly in terms of gauging historical variability and temporal extent and impact of anthropogenic interventions. Fortunately, the accumulation of sediments (both organic and inorganic) within freshwater systems provides records of ecological and physical change and the means to reconstruct environmental history. Developing an enhanced understanding of past system responses to climate changes and

anthropogenic forcing will help ensure future conservation management strategies are underpinned by a strong science base to maximise resilience, preserve diversity of habitats and species, and enhance the natural and sustainable functioning of freshwater aquatic ecosystems.

1.2 Wetland ecosystems

Wetlands are areas where water is the primary factor controlling the environment and associated plant and animal life (Ramsar, 2009), but they are transition zones that reflect the close presence of the water table, which might lie at, near or just above the land surface. Set at the interface between terrestrial and open water systems, wetlands contain features of both and, as such, are highly variable in morphology, ecology and hydrology; but they also exhibit two characteristics that make them wholly unique (Van der Valk, 2012). First, their anaerobic soils, which develop from depleted dissolved oxygen levels due to the abundance of microorganisms in water-saturated soils, is a limiting factor for many plants (Wheeler & Procter, 2000). As a result, the majority of plants and organisms that live in wetlands must have anatomical, morphological, physiological or behavioural adaptations to enable them to survive in such saturated environments. The close proximity of aerobic and anaerobic environments in wetlands is what makes them and their flora and fauna unique (Van der Valk, 2012). Second, wetlands can be clearly distinguished from other ecosystems due to the abundance of large primary producers, namely macrophytes. Macrophytes and their litter influence much of the wetland's physical structure and they modify environmental conditions within the water such as temperature, velocity and chemistry. These large plants have growth forms similar in structure and function to terrestrial trees, shrubs, ferns and mosses; however their underwater adaptations mean they are an inherently different group of plants altogether (Van der Valk, 2012).

Wetlands are found across the globe, from saline coastal areas to continental interiors, and under the full range of climatic conditions, from deserts to tropical rainforests. Three key factors (flooding, disturbance and nutrients) control much of the variation in wetland communities. To understand and manage wetlands, it is essential to appreciate their multifactoral and dynamic nature where specific communities or habitats are produced by multiple environmental factors acting simultaneously (Figure 1.1). Wetlands are vital life-supporting systems that provide a multitude of services contributing to human well-being and poverty alleviation (Millennium Ecosystem Assessment, 2005; UK National Ecosystem Assessment, 2011). *“Wetlands are among the most important ecosystems on Earth”* (Mitsch & Gosselink, 2007: 3). Since early civilisation, many cultures have learned to live in harmony with wetlands and have benefitted economically from surrounding wetlands (Singh, 2010). This is in part due to the role wetlands play as sources, sinks and transformers of a multitude of biological, chemical and genetic materials; providing water as a resource, carbon sequestration and the transformation of nutrients, organic compounds, metals, and organic matter components.

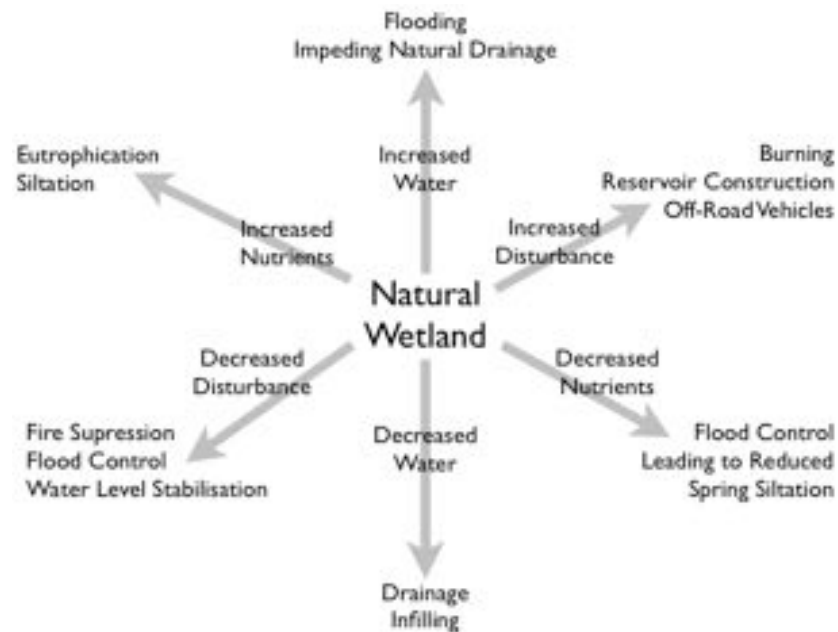


Figure 1.1 Environmental factors determining wetland community and ecosystem type (adapted from Keddy, 2010; p31).

Wetlands are often described as the ‘kidneys of the landscape’ filtering water and waste from both natural and human upstream sources, and they also stabilise water supplies by ameliorating impacts from drought and floods. They have been described as ‘ecological supermarkets’ due to the wide array of habitats that they provide and their consequent rich biodiversity (Mitsch & Gosselink, 2007). Related to this, they are being increasingly recognised as important carbon sinks and hence climate stabilisers on a global scale. This has helped contribute to increasing worldwide recognition of wetlands and the rise of wetland conservation, protection laws, regulations and management plans (Dugan, 1993; Barbier et al, 1997; Cox & Campbell, 1997; Darras et al, 1999; Finlayson et al, 1999; Ramsar Convention Secretariat, 2004; Millennium Ecosystem Assessment, 2005; Fernandex-Prieto et al, 2006; Erwin, 2009; Ramsar, 2009; Maltby & Acreman, 2011; UK National Ecosystem Assessment, 2011). Through increasing awareness and appreciation of wetlands, a need to define and classify these systems has followed to streamline efforts to protect them for future generations (Semeniuk & Semeniuk, 1995; Wheeler & Shaw, 1995; Warner & Rubec, 1997; Allott et al, 2001; Hakala, 2004; Martin et al, 2011).

1.2.1 Wetland definitions and classifications

More than 50 definitions of wetlands are used throughout the world, created by scientists, politicians, lawyers, managers, conservationists, to name but a few. For example, Keddy (2010) describes wetlands as ecosystems that arise when inundation by water (cause) produces soils dominated by anaerobic processes (proximate effect), which in turn forces the biota, particularly rooted plants, to adapt to flooding (secondary effect). In contrast, the United States Committee on Characterisation of Wetlands (1995) define wetlands as *“the minimum essential characteristics of a wetland are recurrent, sustained inundation or*

saturation at or near the surface and the presence of physical, chemical and biological features reflective of recurrent, sustained inundation or saturation. Common diagnostic features of wetlands are hydric soils and hydrophytic vegetation" (Keddy, 2010; p2). The plethora of wetland definitions is due, in part, to the great range of wetland contexts and geographies and also a product of their dynamic nature and difficulties determining boundaries between aquatic and terrestrial ecosystems (Van der Valk, 2012). Nevertheless, Van der Valk (2012) notes that despite some differences in the wording and the emphasis of the various wetland definitions, most are surprisingly consistent.

The Ramsar Convention 'on Wetlands of International Importance especially as Waterfowl Habitat' is an intergovernmental treaty for wetland conservation, adopted in 1971 in the city of Ramsar, Iran. The Convention has to date, gathered 169 contracting parties designed to provide international protection to the widest possible group of wetland ecosystems (Ramsar, 2016). The Convention uses a very general definition of wetlands including swamps, marshes, lakes, rivers, wet grasslands, peatlands, oases, estuaries, deltas, tidal flats, near-shore marine areas, mangroves, coral reefs and man-made sites such as fish ponds, rice paddies, reservoirs and salt pans (Dugan, 2005). The Convention defines wetlands very broadly and concisely in Article 1 (Ramsar, 2009, p1) as:

"Areas of marsh, fen, peatland or water, whether natural, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine waters the depth of which at low tide does not exceed six meters."

And included a broader definition in Article 2:

"May incorporate riparian zones and coastal zones adjacent to wetlands and islands or bodies of marine water deeper than six meters at low tide lying within the wetlands."

With increased need for clarity within the definition, particularly in the context of management and conservation related legislation, the Committee on Characterisation of Wetlands developed a reference definition that ensures all wetlands are covered and hence legally protected (Keddy, 2010: 2):

"A wetland is an ecosystem that depends on constant or recurrent, shallow inundation or saturation at or near the surface of the substrate. The minimum essential characteristics of a wetland are recurrent, sustained inundation or saturation at or near the surface and a presence of physical, chemical, and biological features reflective of recurrent, sustained inundation or saturation. Common diagnostic features of wetlands are hydric soils and hydrophytic vegetation. These features will be present except where specific physiochemical, biotic, or anthropogenic factors have removed them or prevented their development."

The peatland inventories of Europe and North America in the early 1900s started a trend in the classification of wetlands, and the US has since built two major classification schemes on the back of these early inventories (Mitsch & Gosselink, 2007). Spurred on by

recognition of the value of wetlands, the main purpose of such classifications is to increase protection. To achieve this, the primary objectives of wetland classification are therefore “to impose boundaries on wetland ecosystems for the purposes of inventory, evaluation and management” (Cowardin et al, 1979). To deal realistically with wetlands on a regional or even national scale, managers have found it necessary to categorise wetlands based on their type and to determine their extent and distribution (Mitsch & Gosselink, 2007). Possibly the simplest classification systems are those described by Mitsch & Gosselink (2007) and Keddy (2010) who categorise wetlands based on their overall ecohydrology and location, the former categorising wetlands into either ‘coastal’ or ‘inland’, and in the case of the latter, grouping them into swamps, marshes, fens, bogs, wet meadows or shallow water (Table 1.1).

Table 1.1 Simple wetland classification system based on ecohydrological and locational characteristics (adapted from Keddy, 2010; p18).

Wetland	Ecohydrology	Examples of Locational Characteristics
Swamp	Dominated by trees rooted in hydric soils, but not in peat	Tropical mangrove swamps of Bangladesh, bottomland forests in the Mississippi river floodplains
Marsh	Dominated by herbaceous plants that are usually emergent through water and rooted in hydric soils, but not in peat	Cattail <i>Typha angustifolia</i> marshes around the Great Lakes and reed <i>Phragmites australis</i> beds around the Baltic Sea
Fen	Dominated by <i>Sphagnum</i> moss, sedges, ericaceous shrubs, or evergreen trees rooted in deep peat with a pH of less than 5	Blanket bogs of mountainous northern Europe and peatland of west Siberian Lowland, Central Russia
Bog	Dominated by sedges and grasses rooted in shallow peat, often with considerable groundwater movement, with pH greater than 6. Many occur on calcareous rocks and most have brown mosses, in genera including <i>Scorpidium</i> or <i>Drepanocladus</i>	Peatlands of northern Canada and Russia, and smaller seepage areas throughout the temperate zone
Wet Meadow	Dominated by herbaceous plants rooted in occasionally flooded soils. Temporary flooding excludes terrestrial plants and swamp plants from colonising, but drier growing seasons then produce plant communities typical of moist soils	Wet prairies along river floodplains and herbaceous meadows on shorelines of large lakes
Shallow Water	Dominated by truly aquatic plants growing in and covered by at least 25cm of water	Littoral zones of lakes, bays in rivers, more permanently flooded areas of prairie potholes

There are significant limitations in attempting to sort the diversity of nature into only a handful of categories, as wetlands show greater variation than such simple systems can accurately portray. As a result, Mitsch and Gosselink (2007) stress that classifications are only valuable if the user is familiar with their scope and limitations. Cowardin & Golet (1995) warn that “no single system can accurately portray the diversity of wetland conditions world-wide. Some important ecological information inevitably will be lost through classification”. Nevertheless, attempts have been made to improve the accuracy of wetland classification by developing more complex classification systems. More detailed and therefore more focussed classification systems attempt to reduce variability within classes caused by differences in local geology, hydrology, chemistry and climate (Singh, 2010). For example, principal kinds of wetlands can be related to sets of environmental factors. As Figure 1.2 shows, Gopal et al (1990) have categorised wetlands based on a

structure defined by water regime and nutrient supply. Classifications have also been based on the relative proportions of precipitation, groundwater and overland flow (Keddy, 2010). Attempts have also been made to incorporate coastal wetlands, as illustrated in the Cowardin et al (1979) classification of wetlands and deepwater habitats (Figure 1.3). Cowardin et al (1979) also state that the type of classification system chosen depends on the particular scientific, management and regulatory application of interest.

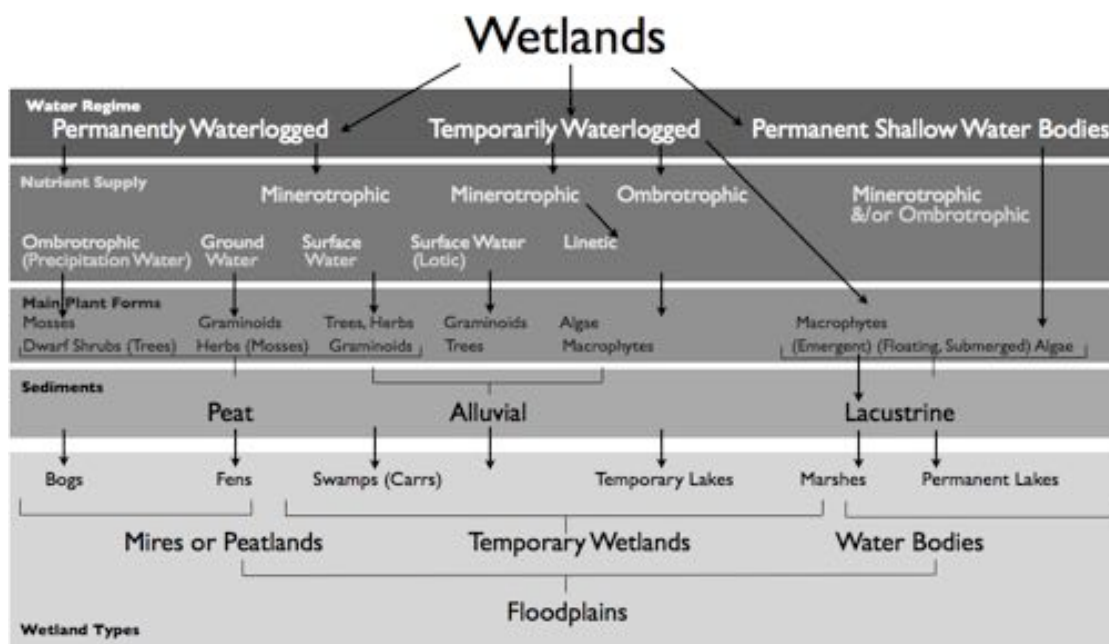


Figure 1.2 Wetland classification based on two sets of environmental factors: water regime and nutrient supply (adapted from Gopal et al, 1990; cited in Keddy, 2010; p36).

The Ramsar Convention classification system is widely used and adapted to form the basis of some national wetland classifications (Table 1.2). For example, the Australian and New Zealand Environment and Conservation Council (aNZECC) adopted the Ramsar Convention classification system in 1994 and adapted it slightly by adding non-tidal freshwater forested wetlands and rock pool categories (Singh, 2010). More complex classifications, not based on the Ramsar Convention, include the Tropical Caribbean System, which is broad enough to encompass everything from mangal to montane seeps and simple enough to require information on only four main criteria (Keddy, 2010). The classification system breaks down the huge diversity of Caribbean wetlands into a number of ecosystem types. Diversity in the Caribbean is due to excess of rainfall over varied topography; therefore this classification system is largely based on geology and hydrology. In contrast, Thompson & Hamilton (1983) recognise four main African wetland types and subdivide them based on location and dominant floral species. In Europe, wetland classification systems are refined to finer and finer scales where each vegetation community type is given a separate name (Keddy, 2010).

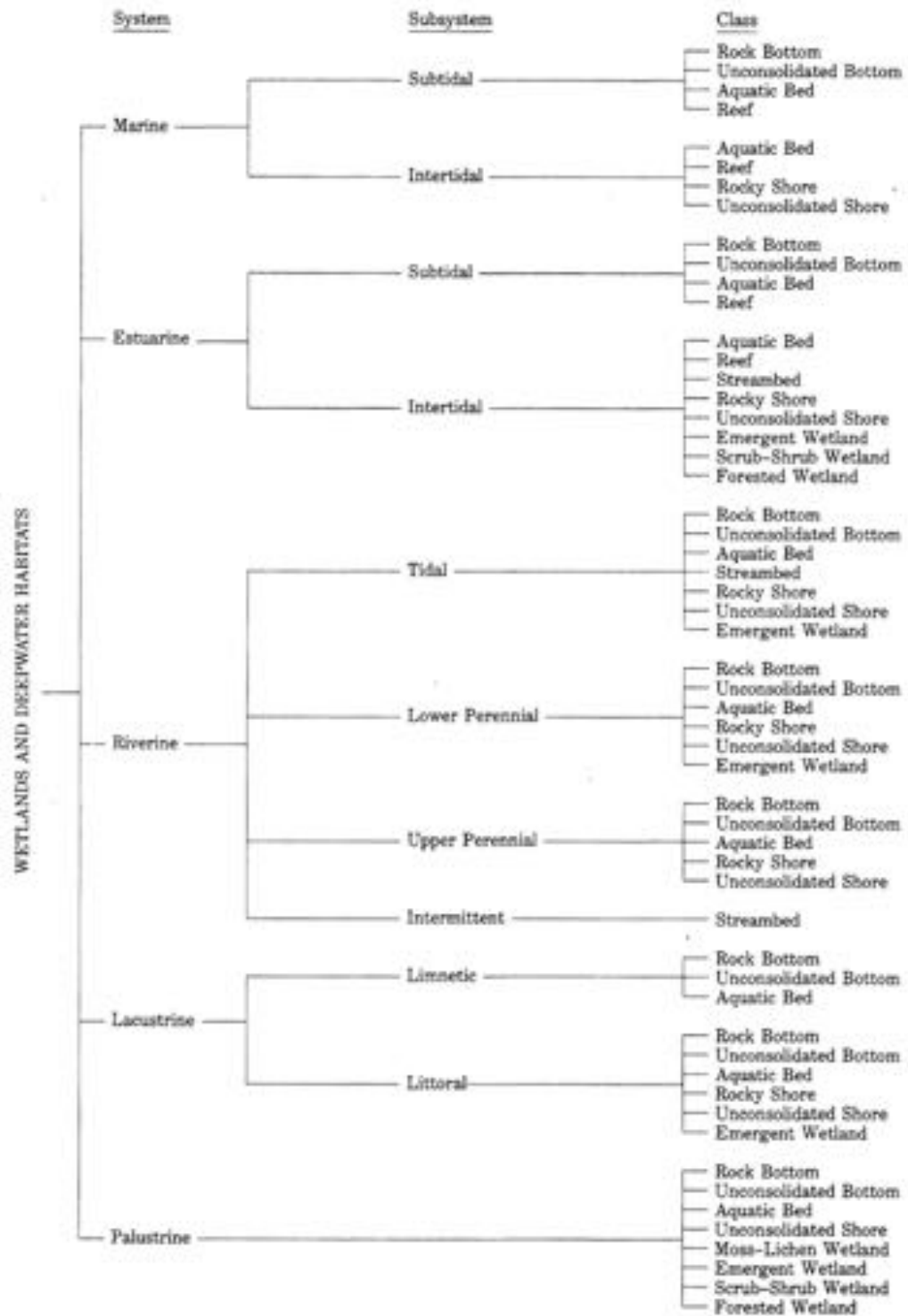


Figure 1.3 Classification tree of wetlands and deepwater habitats (sourced from Cowardin et al, 1979; p5).

Table 1.2 Ramsar Convention classification system of wetland type (Ramsar, 2009).

Marine/Coastal Wetlands	
A	Permanent shallow marine waters less than six metres deep at low tide; includes sea bays and straits.
B	Marine subtidal aquatic beds; includes kelp beds, sea-grass beds, tropical marine meadows.
C	Coral reefs.
D	Rocky marine shores; includes rocky offshore islands, sea cliffs.
E	Sand, shingle or pebble shores; includes sand bars, spits and sandy islets; includes dune systems.
F	Estuarine waters; permanent water of estuaries and estuarine systems of deltas.
G	Intertidal mud, sand or salt flats.
H	Intertidal marshes; includes salt marshes, salt meadows, saltings, raised salt marshes; includes tidal brackish and freshwater marshes.
I	Intertidal forested wetlands; includes mangrove swamps, nipah swamps and tidal freshwater swamp forests.
J	Coastal brackish/saline lagoons; brackish to saline lagoons with at least one relatively narrow connection to the sea.
K	Coastal freshwater lagoons; includes freshwater delta lagoons.
Inland Wetlands	
L	Permanent inland deltas.
M	Permanent rivers/streams/creeks; includes waterfalls.
N	Seasonal/intermittent/irregular rivers/streams/creeks.
O	Permanent freshwater lakes (over 8 ha); includes large oxbow lakes.
P	Seasonal/intermittent freshwater lakes (over 8 ha); includes floodplain lakes.
Q	Permanent saline/brackish/alkaline lakes.
R	Seasonal/intermittent saline/brackish/alkaline lakes and flats.*
Sp	Permanent saline/brackish/alkaline marshes/pools.
Ss	Seasonal/intermittent saline/brackish/alkaline marshes/ pools.*
Tp	Permanent freshwater marshes/pools; ponds (below 8 ha), marshes and swamps on inorganic soils; with emergent vegetation water-logged for at least most of the growing season.
Ts	Seasonal/intermittent freshwater marshes/pools on inorganic soil; includes sloughs, potholes, seasonally flooded meadows, sedge marshes.*
U	Non-forested peatlands; includes shrub or open bogs, swamps, fens.
Va	Alpine wetlands; includes alpine meadows, temporary waters from snowmelt.
Vt	Tundra wetlands; includes tundra pools, temporary waters from snowmelt.
W	Shrub-dominated wetlands; Shrub swamps, shrub-dominated freshwater marsh, shrub carr, alder thicket; on inorganic soils.*
Xf	Freshwater, tree-dominated wetlands; includes freshwater swamp forest, seasonally flooded forest, wooded swamps; on inorganic soils.*
Xp	Forested peatlands; peat swamp forest.*
Y	Freshwater springs; oases.
Zg	Geothermal wetlands.
Zk	Subterranean karst and cave hydrological systems.
<i>* Also includes as appropriate: floodplain wetlands such as seasonally inundated grassland (including natural wet meadows), shrublands, woodlands or forest.</i>	
"Man-made" Wetlands	
1	Aquaculture (e.g. fish/shrimp) ponds.
2	Ponds; includes farm ponds, stock ponds, small tanks; (generally below 8 ha).
3	Irrigated land; includes irrigation channels and rice fields.
4	Seasonally flooded agricultural land.**
5	Salt exploitation sites; salt pans, salines, etc.
6	Water storage areas; reservoirs/barrages/dams/impoundments; (generally over 8 ha).
7	Excavations; gravel/brick/clay pits; borrow pits, mining pools.
8	Wastewater treatment areas; sewage farms, settling ponds, oxidation basins, etc.
9	Canals and drainage channels, ditches.
<i>** To include intensively managed or grazed wet meadow or pasture.</i>	

1.2.2 Wetland extent and distribution

Wetlands cover about 5% to 8% (7-10 x10⁶km²) of the Earth's surface, and are not restricted to specific regions of the world (Mitsch & Gosselink, 2007). The European Union Water Framework Directive (WFD), which was created in 2000 and committed member states to achieve good qualitative and quantitative ecological status of all water bodies by 2015 (including marine waters up to one nautical mile from shore), has encountered significant difficulties in calculating the global extent of the world's wetlands (Acreman et al, 2007). This is largely due to the fact that no two wetlands are identical and the exact extent of wetlands remains uncertain (Keddy, 2010). Estimates for global extent are varied (Table 1.3) as they depend on definitions, classification methods and quality of

satellite data used (Mitsch & Gosselink, 2007). For example, Finlayson et al (1999) based their global wetland estimate of $12.8 \times 10^6 \text{ km}^2$ on the definition created by the Ramsar Convention Bureau, which includes inland and coastal wetlands, near shore marine areas and man-made wetlands such as reservoirs and rice paddies. By basing the estimate on this definition, however, wetland types such as flooded inland wetlands, peatlands, artificial wetlands, seagrasses and coastal flats are all under-represented. While definitions remain extremely varied, more accurate estimates of global wetland area will not be possible.

Table 1.3 Comparisons of the estimated extent of global wetlands by climatic zone (adapted from Mitsch & Gosselink, 2007; p46).

Zone	Wetland Area ($\times 10^6 \text{ km}^2$)						
	Maltby & Turner (1983)	Matthews & Fung (1987)	Aselmann & Crutzen (1989)	Gorham (1991)	Finlayson & Davidson (1999)	Ramsar Convention (2004)	Lehner & Doll (2004)
Polar/Boreal	2.8	2.7	2.4	3.5	-	-	-
Temperate	1.0	0.7	1.1	-	-	-	-
Subtropical/Tropical	4.8	1.9	2.1	-	-	-	-
Rice Paddies	-	1.5	1.3	-	-	1.3	-
Total Wetland Area	8.6	6.8	6.9	-	12.8	7.2	8.2-10.1

1.2.3 Threats to wetlands

Throughout history and in particular since 1800, wetlands have been drained, ditched and filled. Wetland loss is often viewed as a key representation of the overall global ecological degradation and loss of natural ecosystems to economic development. Despite significant and obvious losses, it is impossible to accurately quantify the impact that humans have had on wetlands over the centuries. What can be said, however, is that in developed and heavily populated regions of the world, the impact of man on wetlands is significant (Mitsch & Gosselink, 2007). Since early civilisation, wetlands have become increasingly negatively exploited and many have disappeared as a result of agricultural and urban development (Maltby & Acreman, 2011). Major cities all around the world including Chicago and Washington DC in the United States, Christchurch in New Zealand, Paris in France and London in England were built on sites that encompass former wetlands (Mitsch & Gosselink, 2007). It is through the control of nature that wetlands worldwide have met their demise (Mitsch & Gosselink, 2007).

Despite their worldwide distribution and the increasing efforts in wetland conservation and management, wetland extent and distribution continues to decline in some regions. Threats to wetlands have been commonplace for centuries, with many wetlands having been completely drained hundreds of years ago, and hydrological modification via dykes, dams and levees continue to present significant threats to associated ecosystems (Mitsch & Gosselink, 2007). Wetland loss has been attributed to drainage for agriculture and

forestry, filling for residential, commercial and industrial development and mining for peat (Maltby & Acreman, 2011). In particular, western and central Europe, which contains some of the most densely industrialised and intensively farmed landscapes in the world, has lost the vast majority of its natural wetlands (Mitsch & Gosselink, 2007).

In the wake of centuries of agricultural and industrial revolutions and more recent economic development, the wetlands of western and central Europe have been reduced to small isolated pockets on lowland plains and in the high Alps (Dugan, 2005). Nevertheless, the region still contains some of the most productive and biologically important systems in the world, most notably the estuaries and coastlines of western Europe, the floodplains of major rivers, and the wetland mosaics of the central European plain (Dugan, 2005). The primary direct drivers of degradation and loss of wetlands include infrastructure development, land conversion, withdrawal of water, pollution, introduction of invasive species, overharvesting and overexploitation (Millennium Ecosystem Assessment, 2005). Wetlands depend on the maintenance of natural hydrological conditions in order to preserve their biodiversity, functions and values (Gopal, 2009). Human modification of hydrological conditions by, for example, removing water or altering processes through abstraction, artificial storage or diversion via public water supplies, can have hugely detrimental consequences for wetlands. For example, clearing and drainage (often for agricultural expansion), are the main reasons for the loss and degradation of inland wetlands such as swamps, marshes, rivers, and associated floodplain water bodies (Mitsch & Gosselink, 2007). By 1985, an estimated 56–65% of inland and coastal marshes (including small lakes and ponds) had been drained for intensive agriculture in Europe and North America, 27% in Asia, 6% in South America, and 2% in Africa (Millennium Ecosystem Assessment, 2005). The extensive use of water for irrigation (approximately 70% of water is used globally for irrigation) has resulted in a decline in freshwater ecosystem services. In addition, rapid and unsustainable development within wetlands and their surrounding catchments has led to significant disruptions of hydrological conditions, including frequency and severity of flooding events, as well as enhanced incidences of droughts and pollution (McCartney & Acreman, 2009).

According to the Intergovernmental Panel on Climate Change (IPCC) predictions relating to intensification of the hydrological cycle due to rising global temperatures, the greatest impact of climate change upon wetlands will be to their hydrological regimes (IPCC, 2007; Erwin, 2009; Baker et al, 2009). More specifically, temporal and spatial patterns of water levels will be altered, as will the magnitude of drought and flood extremes. It should be pointed out, however, that the nature and magnitude of climate change impacts will vary between wetland types and locations. For many wetlands, the most significant impacts are likely to be associated with changes in the amount, state and seasonal distribution of precipitation, higher evaporation due to warmer temperatures and the combined effects of these upon runoff (Hartig et al, 1997; Mortsch, 1998; Conly & van Kamp, 2001). Freshwater wetlands tend to be particularly vulnerable to climate change induced modifications to hydrological regimes due to a delicate balance between precipitation and evaporation (Clair, 1998; Thompson et al, 2009). In addition, hydrological changes will

impact the flora and fauna within and around wetlands due to their sensitive water level preferences (Mortsch, 1998).

Although they cover approximately six percent of the planet's surface, wetlands account for more than ten percent of all animal species and thirty-five percent of all vertebrate species (Stendera et al, 2012). For example, about 29,000 species have been identified in freshwaters, comprising 12,000 fish species, with the remainder including diverse groups such as microorganisms, algae, insects, crustaceans, annelids and molluscs (Palmer et al, 2000). Decline in freshwater biodiversity, however, is exceeding that of other ecosystems (Dudgeon et al, 2006) and human pressures on freshwaters and other wetland systems have increased tremendously over the last century. Wetlands are now hot spots of both biodiversity and species endangerment around the world as a result of habitat destruction, fragmentation, invasive species, pollution, human population growth and overharvesting (Dudgeon et al, 2006). Currently, about 2,251 species (41%) of the 5,435 animals listed by the 2000 IUCN Red List are living in aquatic environments (IUCN, 2011). Rare plants are particularly at risk as it has been estimated that nearly one third of threatened and endangered plant species depend on wetlands for survival (Niering, 1988; Murdock, 1994). This in turn could alter the conservation significance of some designated sites (Burkett & Kusler, 2000; Herron et al, 2002; Bates et al, 2008; Matthews & Quesne, 2009). Biological and biogeochemical functions within the wider wetland ecosystem will also be affected, which together will affect the socio-economic benefits that are valued so highly, either consciously or unconsciously, by human civilisation (Cox & Campbell, 1997).

1.2.4 Wetland conservation management

The concept of wetland management has varied in meaning across the world and throughout history. Until the mid twentieth century, most policy makers viewed wetland management as 'wetland drainage', except those resource managers who promoted more traditional activities such as hunting and fishing, and more recently, the introduction of wildfowl and wildlife protection (Mitsch & Gosselink, 2007). Up until that time landowners were encouraged by government legislation to drain wetlands in order to make more land suitable for agriculture, whilst a large number were filled to accommodate development. Today, an increase in understanding of the importance of wetlands plus greater recognition of the vital functions and services that they provide has led to wetland management rising up the political agenda. Wetlands are now recognised across the globe as habitats requiring sustainable conservation management. Maltby & Acreman (2011) illustrate this paradigm shift in a conceptual model, which highlights the central role that wetlands now play in influencing society, science and management (Figure 1.4).

Government policy in a multitude of countries now seeks to reverse historic wetland losses in the face of continuing drainage or encroachment by agricultural enterprises and urban expansion. This task is largely based on the Ramsar Convention mission statement (Ramsar, 2009): *"the conservation and wise use of all wetlands through local, regional and national actions and international cooperation, as a contribution towards achieving*

sustainable development throughout the world." As a result of the Ramsar Convention, and other key milestones in the evolution of wetland science and policies (for example the Clean Water Act in the USA and the European Water Framework Directive), many countries and nongovernmental organisations (NGOs) have begun to dedicate policies and resources to preserving wetlands (Smith et al, 1995; Brinson, 2009; Maltby, 2009; Singh, 2010).

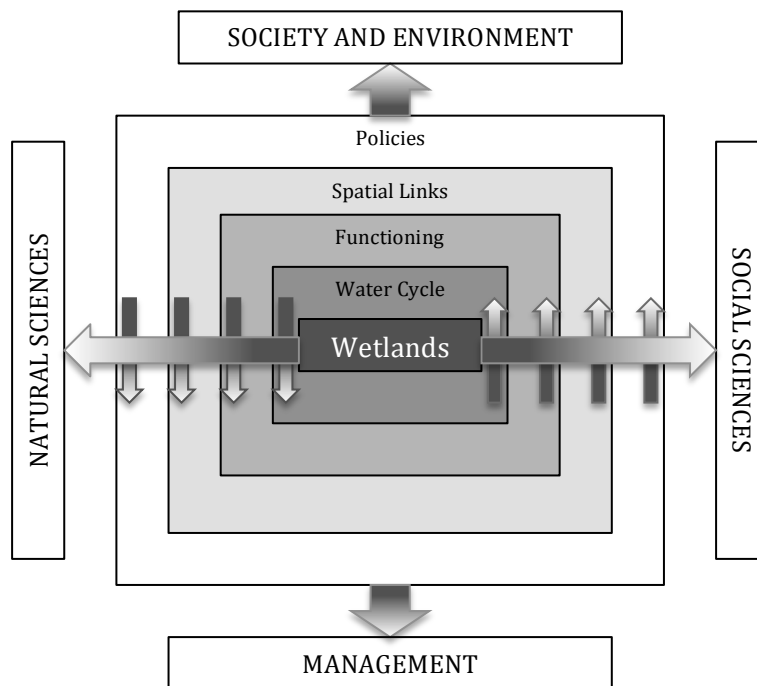


Figure 1.4 Conceptual model showing wetlands at the centre of concerns of society (adapted from Maltby & Acreman, 2011; p1342).

Wetland management is a balancing act that requires setting objectives based on the priorities outlined by wetland managers, current environmental regulations and wishes of the myriad of stakeholders involved (Mitsch & Gosselink, 2007). Wetland management has long relied on the strength and significance of the conservation lobby, a relationship that will no doubt continue to be important (Maltby & Acreman, 2011). In the face of global population increase and consequent rising world food and commodity prices, however, there is growing concern in meeting basic human needs, such as clean water, relief from poverty and safety from environmental hazards. Maltby & Acreman (2011) stress that there is therefore a growing need for a human-centric approach to wetland management that could be structured around ecosystem services, e.g. cleaner water, reduced flooding, slower climate change, improved human health. They recommend that successful implementation of such an approach would require widespread buy-in from vested interests, especially farming communities. Innovative tools such as payments for ecosystem services could then be used to reduce the imbalances between humans and nature.

1.3 Lake ecosystems

Forel defined lakes as *“a body of standing water occupying a basin and lacking continuity with the sea”* (Forel, 1901; p253), while more recent definitions define lakes as *“large or considerable bodies of standing water either salt or fresh surrounded by land”* (Timms, 1992; p2). Shallow lakes, which are considered by many authors to be wetlands and not lakes, are described by Löffler (2004) as bodies of water that are easily mixed down to their base by wind, evaporation or irradiation, and are dependent on location. Conversely, others argue that periods of stratification can be induced even in the shallowest of lakes by factors such as ice cover, freshwater inflows into saline lakes, or by dense floating vegetation that may influence fetch and, less so, evaporation (Scheffer, 1998; Padisak & Reynolds, 2003).

Lake definitions continue to divide opinion and, despite the progress made by Forel and those who came after him, it was only until the early twentieth century, that lakes became a focal unit in the biological study of natural system functioning (Reynolds, 2000). Landscape ecologist, August Thienemann, realised the distinctive properties of lakes and their crucial interaction with surrounding catchments (Thienemann, 1925), while Macan (1979) recognised that the study of lakes must include some appreciation of their developmental history. Since those pioneering studies, experts have come to realise that the biology of lakes is influenced by a significant range of factors including location, physiography, climate, chemistry, hydrology, morphometry and the adaptations of aquatic biota (Reynolds, 2004). They have also learnt that no two water bodies will ever be identical. Despite this knowledge, the search for underlying environmental patterns and processes continues, moving us towards a better understanding of the ecology of lakes and their biota.

Lakes have long been the focus of studies of ecological processes such as community assembly, competitive exclusion and niche differentiation, largely because such processes occur over shorter timescales (seconds to decades) as opposed to many terrestrial processes that require years to millennia of study to reach full understanding (Birge & Juday, 1934; Odum & Barrett, 1971; Reynolds, 2004). Fascination for the island nature of lakes and their existence as ‘patches’ of suitable habitat in otherwise inhospitable terrestrial environments, have perhaps permitted limnologists to make significant advances in developing realistic models of species interactions with the environment (Birge & Juday, 1934; Steel, 1995; Reynolds & Irish, 1997). In addition, studies have begun to recognise that the presence of fluvial systems in conjunction with lakes significantly influences the viability and survival of lake species, therefore highlighting the importance of obtaining thorough ecological understanding of these systems (Tokeshi, 1994).

1.3.1 Lake classification

Lake ecosystems are inherently variable in their physical, chemical and biological properties (Emmons et al, 1999). The origin of a lake, its geology, morphology, climate, geochemistry and hydrology all contribute to its unique characteristics which together

influence biological processes. Although the relative importance of these factors is debatable and variable among lakes, Rawson (1939), Deevey (1940), and Mortimer (1942) are unanimous in their acknowledgement of their combined importance (Carpenter, 1983). The uniqueness of lakes and the biota they support presents a challenge for conservation managers. For conservation to be effective over different spatial and temporal scales, managers require strategies founded upon a lake classification system that has both a simplified structure and also the capacity to accommodate characteristics of individual lakes. Researchers have been classifying lakes since the early 1900s, with lake classification the major focus of the International Congress for Limnology in 1956 (Martin et al, 2011). Classification systems that allow for ecological, hydrological and geological variation provide a way to group lakes into units in which similar responses to anthropogenic impacts or management strategies can be applied. Classification systems that do this effectively can also provide a framework for more structured research into the ecological functioning of lake ecosystems.

The difficulties that arise in the wetland classification process are also apparent in lake classification. Indeed, many lake classification schemes have been proposed, but most are never wholly satisfactory (Lewis, 1983). Only two classifications are used frequently: the trophic classifications of Naumann (1919) and Hutchinson (1973), and the circulation classification of Hutchinson and Löffler (1956). These two classification systems have most likely persisted due to their simplicity and because they are based on two of the most fundamental means for comparing lakes: nutrition and temporal variation (Lewis, 1983). As lakes occur in a multitude of environments across the globe and lake formation is extremely varied, occurring over wide-ranging, but often extremely long time periods, lake classification is often considered in terms of decreasing temporal scales: from centuries and millennia (macroscale), to decades and centuries (mesoscale), and to months and years (microscale).

1.3.1.1 Macroscale: centuries to millennia

At the macroscale (centuries to millennia) classification is focussed on the origin (genesis), development and morphology of lakes. Davis in 1882 explored lakes in the context of geological processes and divided them into constructive (e.g. volcanic crater lakes), destructive (e.g. lakes formed by glacial scouring) and obstructive (e.g. lakes generated by natural damming events from processes such as lava flows and landslides). Similar classification systems were developed by Penck (1894), Forel (1901) and Halbfass (1923); however it was not until 1957 that Hutchinson developed a far more comprehensive system that included 76 lake sub-types grouped under 11 major kinds of formation which include: tectonic, volcanic, glacial, solution, fluvial, coastal and dammed lake types (Table 1.4) (Hutchinson, 1957). The advantage of Hutchinson's classification is it is flexible and can accommodate modifications in the form of additional lake types.

The aim of classifying lakes based on their morphology is based on an attempt to understand the forces that produced lakes and their basins, as well as the events that occur in the drainage basin after formation. Morphological classifications are therefore

rarely used as the sole classification system in lake studies and tend to be regionally or globally focussed (Liu et al, 2010). Accurate morphological classification requires measurement of the planform, watershed topography, lake bathymetry, lake shape (i.e. major/minor axis lengths, area, depth, volume, cryptodepression (the maximum depth of that part or the whole lake lying below sea level)) and the wetted area (Hutchinson, 1957). As Table 1.5 shows, the shape of most lakes reflects their origin and history.

Table 1.4 Lake type based on formative origins defining the Hutchinson (1957) classification system (adapted from Thomas et al, 1996; 326).

Lake Type	Characteristics
Tectonic	<ul style="list-style-type: none"> Formed by large-scale crustal movements separating water bodies from the sea e.g. Aral and Caspian Seas Formed in rift valleys by earth faulting, folding or tilting, such as the African Rift lakes and Lake Baikal, Russia Lakes in this category may be exceptionally old
Volcanic	<ul style="list-style-type: none"> Occur in craters and calderas, including dammed lakes resulting from volcanic activity Common in Japan, Philippines, Indonesia, Uganda, Cameroon, Central America and Western Europe
Glacial	<ul style="list-style-type: none"> On or in ice, ponded by ice or occurring in ice-scraped rock basins Formed by moraines of all types, and kettle lakes occurring in glacial drift The most numerous lake type Occur in all mountain regions, in the sub-arctic regions and on Pleistocene surfaces All of the cold temperate, and many warm temperate, lakes of the world fall in this category e.g. Canada, Russia, Scandinavia, Patagonia and New Zealand
Solution	<ul style="list-style-type: none"> Occur in cavities created by percolating water in water-soluble rocks such as limestone, gypsum or rock salt They are called <i>Karst lakes</i> and are very common in the appropriate geological terrain Considered as small, although there is some evidence that some large water bodies may have originated in this way e.g. Lake Ohrid, Yugoslavia
Fluvial	<ul style="list-style-type: none"> Created by river meanders in flood plains e.g. oxbow and levee lakes Formed by fluvial damming due to sediment deposition by tributaries e.g. delta lakes and meres Formed by geomorphological processes including deflation
Coastal	<ul style="list-style-type: none"> Cut off from the sea by spit creation caused by sediment accretion due to long-shore sediment movement Coastal lakes of Egypt
Dammed	<ul style="list-style-type: none"> Created behind landslides, rockslides, mudflows and screes Short duration but of considerable importance in mountainous regions
Other	<ul style="list-style-type: none"> Excluding reservoirs, many other natural origins for lakes may be defined, ranging from lakes created by beaver dams to lakes formed by plant accumulation, to deflation lakes, and those in meteorite depressions

Correlations between morphology and biological characteristics were first recognised over a century ago (Patalas, 1980). Since then, fish productivity has been related to mean lake depth, and lake morphology has been found to influence productivity by affecting the rate of nutrient recycling from sediments to the water by processes such as diffusion, resuspension, bioturbation and decay of macrophytes (Wetzel, 1979; Kitchell et al, 1979; Carpenter, 1980; De Groot, 1981). Water depth is a primary controller of these biological processes (Table 1.6), but up until relatively recently, lake research largely encompassed the functioning of deep lake ecosystems; however progress is now being made on shallow lake research as their abundance and widespread distribution across the globe has led to greater recognition amongst scientists, policy makers and conservation managers of their value for humans (Cooke et al, 2001).

Table 1.5 Lake classification based on morphology (adapted from Loffler, 2004; p34).

Lake Type	Characteristics
Circular	Formed by volcanic events and meteorite strikes Include explosion craters or maars and crater lakes
Sub-Circular	Formed by volcanic events Include kettle lakes, cirque lakes and pans that have normally been modified by shoreline processes and deflation
Sub-Rectangular Elongate	Formed in grabens, fjords and overdeepened valleys
Lunate	Crescent moon shape Include oxbows, maars and volcanic basins with asymmetrically placed secondary cones
Triangular	Formed by flooding of non-dissected valleys behind bars, sand dunes, spits, levees or artificial dams
Dendritic	Formed in drowned valleys with their lower ends blocked by a dam, mass movement debris or geological tilting
Irregular	Formed in areas where the fusion of basins has occurred

Table 1.6 Characteristics of shallow and deep lakes (adapted from Cooke et al, 2001; p43).

Characteristic	Shallow	Deep
Likely size of drainage area to lake area	High	Lower
Responsiveness to diversion of external P loading	Lower	Higher
Polymictic	Often	Rarely
Benthic-pelagic coupling	High	Low
Internal loading impact on photic zone	High	Low
Impact of benthivorous fish on nutrients/turbidity	High	Low
Fish biomass per unit volume	Higher	Lower
Fish predation on zooplankton	High	Lower
Nutrient control of algal biomass	Lower	Higher
Responsiveness to strong biomanipulation	Higher	Lower
Chance of turbid state with plant removal	Higher	Lower
Probability of fish winterkill	Higher	Lower
Percent area/volume available for rooted plants	High	Low
Impact of birds/snails on lake metabolism	High	Lower
Chance of macrophyte-free clear water	Low	Higher

1.3.1.2 Mesoscale: decades to centuries

At the mesoscale (decades to centuries), lake classification focuses on factors that control medium-term processes such as water properties, hydrology and climate. Three hydrological groups characterise lake watersheds: exorheic, endorheic and arheic watersheds (Table 1.7) (Martonne & Aufrere, 1928; Hutchinson, 1957; Meybeck, 1995). These groups are largely in accordance with the global distribution of arid, semi-arid or semi-humid and humid regions (Loffler, 2004). Although of little use for conservation management strategies at the local scale, hydrological classifications may be useful for providing a broader context for lake conservation. Hydrogeological processes within a catchment are considered in many lake classifications in terms of defining water chemistry properties (Table 1.7) (Bond, 1935; Pitblado et al, 1980; Zimmerman et al, 1983; Young & Stoddard, 1996; Momen & Zehr, 1998; Jenerette et al, 2002; Hakanson, 2005).

Table 1.7 Lake classification based on geochemistry (adapted from Löffler, 2004; p36-46).

Classification	Type	Characteristics
Hydrology	Exhorheic	<ul style="list-style-type: none"> • Empty into the sea via overland flow, usually via a river channel • Represent the major part of the drainage of all the continents except Australia
	Endorheic	<ul style="list-style-type: none"> • Discharge inland, usually into closed lake basins • Mainly restricted to arid and semi-arid regions
	Arheic	<ul style="list-style-type: none"> • No rivers are present • May be crossed by very large river channels • Located in areas such as the lower part of the River Nile, the Oranje, and the Nijer
Chemical Composition	Athalassohaline	<ul style="list-style-type: none"> • Inland salt lakes rich in anions other than chloride or cations other than sodium; magnesium is abundant and occurs in combination with chloride and sulphate • Inland lakes with a wide range of anion and cation compositions
	Thalassohaline	<ul style="list-style-type: none"> • Simulate marine salt composition; sodium chloride is the most abundant ion along with chlorides, bicarbonates, carbonates and sulphates • Bear close hydrological and biological relationships to the sea
Circulation	Holomictic	<ul style="list-style-type: none"> • Complete or partial mixing at least once a year between surface waters and those at depth due to homothermal conditions
	Monomictic	<ul style="list-style-type: none"> • Mixing occurs only once a year
	Dimictic	<ul style="list-style-type: none"> • Mixing occurs twice a year, usually during spring and autumn
	Polymictic	<ul style="list-style-type: none"> • Mixing occurs several times a year
	Meromictic	<ul style="list-style-type: none"> • Layers within the water column remain unmixed for extremely long periods of time, from decades to centuries
Climatology	Amictic	<ul style="list-style-type: none"> • Rare, sealed from any climatic influence by ice • Found in the polar regions
	Cold Monomictic	<ul style="list-style-type: none"> • Mixing only during the summer • Occur in predominantly non-arid sub-polar and extratropical mountain regions, virtually absent from the Southern Hemisphere
	Warm Monomictic	<ul style="list-style-type: none"> • Mixing only during the winter • Found in the Mediterranean region where the climate is strongly influenced by oceans and some sub-tropical mountain ranges in Ireland, New Zealand and most of Japan and southern Chile
	Dimictic	<ul style="list-style-type: none"> • Covered with ice during the winter, mix twice a year during the spring and autumn • Found in temperate regions of the Northern Hemisphere, absent in Ireland; also found in extra-tropical mountain regions
	Cold Polymictic	<ul style="list-style-type: none"> • Cooling during the night at high altitudes results in complete mixing and only weak stratification during the day • Found at high altitude in the tropics, in areas such as the Andes, Mount Kenya and the mountains of Papua New Guinea, where the two annual solar radiation maxima produce a diurnal climate
	Oligomictic	<ul style="list-style-type: none"> • Mix only at very infrequent intervals when extreme cold spells occur • Examples include Lake Nyos in Cameroon and Lake Toba in Sumatra

The geochemistry of a lake is also an important factor in the degree of physical circulation or mixing that occurs within the water column, characteristics that were used by Hutchison (1937) to differentiate between lakes (Table 1.7). Circulation occurs because water density changes with temperature. The original holomictic and meromictic classification was initially based on whether complete mixing had occurred at least once a year or not at all (Findenegg, 1935; Hutchinson, 1937). Nevertheless, the term meromictic later included lakes with irregular circulation and stratification periods. Other terms

developed including 'semi-meromictic', 'temporary meromictic', 'periodic meromictic' and 'spring meromictic' (Hakala, 2004). In addition, Walker & Likens (1975) created the dubious term 'meromictic *sensu lato*' for lakes showing permanent stratification over fifty percent of the time. A more accurate distinction was made by Miracle et al (1993) who coined the term 'extreme meromixis' for meromictic lakes with sharp chemoclines. Another view considers separation between different kinds of holomictic lakes and proposed the term 'incomplete holomixis' for lakes with irregular circulation resulting in fluctuating conditions. This therefore eliminates ambiguity by restricting the term meromixis for meromictic lakes. These conflicting definitions therefore beg the question whether classifying between holomictic lakes with irregular mixing and meromictic lakes is largely impractical and highlights the need for greater clarity within geochemical classifications (Lewis, 1983; Tyler & Vyverman, 1995; Hongve, 2002).

In 1956, Hutchinson & Löffler evolved a new approach to lake classification at the mesoscale that incorporated aspects of circulation (or 'mixing' behaviour of holomictic lakes) in different climatic zones (Table 1.7). This system followed also from Forel's earlier (1895) classification and was designed to take into account latitude, altitude and climate; however its thermal focus means it excludes the classification of shallow lakes (Lewis, 1983). Lewis (1983) suggested revising this classification so that any lake can either be first classified as either meromictic or holomictic and, regardless of which classification is correct, can also be classified accordingly to typology based on seasonal patterns. In addition, meromictic lakes can be assigned a mixing type according to the behaviour of the mixolimnion. The advantages of this revised classification include: lakes being correctly termed meromictic despite not being completely chemically homogenous; the ability to assign an appropriate seasonal mixing category; and making the ambiguous oligomictic term obsolete.

1.3.1.3 Microscale: months to years

At the microscale (months to years), lake typologies focus on short-term climate controls, ecology, conservation and the lake policy context. When climate is viewed over a much shorter timescale, for example over the course of a year, lake classification becomes purely defined on the basis of within-lake characteristics as a function of the seasons. For example, 'holomictic' lakes, which experience partial mixing at least once a year, are deep enough to allow for the development of summer stratification (Löffler, 2004). Lakes in temperate climates exhibiting this seasonal pattern in temperature can hence be classified and distinguished from all other lakes across the globe (Löffler, 2004). In contrast, the 'monomictic' lake type represents lakes that only exhibit summer stratification (Hutchinson & Löffler, 1956). Lakes that exhibit two mixings over the course of a year are classified as 'dimictic'.

Until recently, microscale ecological assessment of lakes has generally relied on abiotic measurement (Allott et al, 2001; White & Irvine, 2003). For example, the impacts of acidification on lake ecology has largely been measured using pH and acid neutralizing capacity (OECD, 1982; Henriksen et al, 1992; White & Irvine, 2003). Lake trophic status

has thus been used to classify lakes using measurements of nitrogen, phosphorus and chlorophyll a. In 'oligotrophic' lakes, for instance, macrophyte and phytoplankton growth is severely limited by very low nutrient levels in the water column (Pokorny & Kvet, 2004). Macrophytes capable of acquiring nutrients through their roots monopolise an alternative resource opportunity; however transparency remains high due to the lack of suspended biota leading to high biomass but low productivity. In moderately nutrient rich 'mesotrophic' lakes, aquatic biota is more abundant and diverse, as nutrient availability remains constant throughout the year (Pokorny & Kvet, 2004). Rooted plants are evenly distributed in the euphotic zone, although submerged macrophyte shoots only rarely reach the water surface except for those species that develop floating leaves. In 'eutrophic' lakes high nutrient loading, whether naturally or anthropogenically induced, leads to significant plant growth and a greater standing biomass (Pokorny & Kvet, 2004). Vigorous growth results in most cases in biomass accumulating throughout the water column and at the water surface. This biomass accumulation shades deeper regions, reducing transparency and growth on the lakebed.

The use of biotic variables to assess lake condition is rarer, however, with profundal and sub-littoral invertebrate communities used as supporting indicators of trophic status, acidity and heavy metal content (Wiederholm, 1980; Fjellheim & Raddum, 1990; Reynoldson et al, 1995; Lang & Reymond, 1996). Naumann (1919) classified temperate lakes on the basis of their plant production and was primarily concerned with the visual signs of eutrophication. Such studies have therefore highlighted a number of gaps in the knowledge required for effective assessment of lake ecology (Murphy et al, 2002). Nevertheless, the lack of understanding of lake ecology is rather unsurprising, especially when considering the inherent heterogeneity of limnological systems (Rasmussen, 1988; Johnes et al, 1996; Harrison & Hildrew, 1998; Koskenniemi, 2000).

Despite this heterogeneity, there is still a tendency among scientists and conservation managers to classify lakes and group them along the lines of commonality. The European Water Framework Directive (WFD), adopted by the European Union in December 2000 to protect and improve the quality of all surface water resources, operates using five different ecological classes assessed by using a wide variety of biotic variables including phytoplankton, macrophytes, invertebrates and fish (European Union, 2000; Sondergaard et al, 2005). These classes are rather vague in their specification, however, only providing general guidance on how to define target water bodies. Indeed, Loffler (2004) argues that the true limnologist needs to see the wider picture to recognise the interactions among adjacent yet distinct habitats within a single lake and the ways in which their relative magnitudes may define many of the differences between lakes. In order to achieve a better understanding of lake ecosystems, ecological classification systems have therefore been developed to identify important organisms or groups of organisms that dominate the food web and have the greatest influence over the flow of energy and matter through a lake ecosystem (Allott et al, 2001). It has also been recognised that these classifications should be based on an adequate representation of the range of variation in lake ecosystem types for the area in question (NCC, 1989; Sætersdal & Birks, 1993; Duigan & Kovach, 1994).

An integrated view of lake ecosystems is therefore emerging, largely in response to the

need to understand and manage the behaviour of whole lake systems for conservation. Site evaluation is a key aspect of conservation practice; therefore criteria for evaluating lakes of conservation interest now include broader factors such as species and habitat diversity, the area of the site, rarity of species or habitats, naturalness and representivity (Usher, 1986). Classifications are evolving to include consideration of all relevant biological and environmental attributes of lake ecosystems (Duigan & Kovach, 1994; Lassiere & Duncan, 1997). Lake conservation is therefore moving away from evaluations based on individual species or indicator groups towards approaches which aim to include a more complete expression of site resources (Usher, 1986). There is also a growing awareness in lake conservation of the importance of site history to conservation evaluation and restoration (Birks, 1996). Recent studies emphasising the dynamic nature of lakes have demonstrated that many lake ecosystems have been significantly impacted by anthropogenic activity, especially by nutrient enrichment and acid deposition (Battarbee et al, 1988, Bennion, 1994, Battarbee, 1999). Eutrophication and acidification affect both lake water chemistry and aquatic biota, and their dynamic nature means that current conditions do not necessarily reflect the true status of the lake or the potential for change. These findings have therefore lead to criticism of spatial-state classification schemes that ignore short-term dynamics (Moss et al, 1994, Moss et al, 1996; Battarbee, 1997; Battarbee, 1999; Fozzard et al, 1999). As a result, lake status is increasingly being defined by reconstructing the pre-anthropogenic state or reference condition using palaeolimnological techniques (Allott et al, 2001). The present state of a lake is compared to the reference state to evaluate the degree of ecosystem change and anthropogenic impact, meaning they can be classified on the basis of both reference condition and degree of change (Allott et al, 2001). By selecting lakes for conservation based on consideration of the full range of ecosystem parameters, scientists and managers hope that the uniqueness of different lakes, when not clearly displayed by their geology, hydrology and geochemistry, can be more easily identified by their ecology. Conservation strategies can therefore be tailored accordingly to individual lakes of similar abiotic character whilst also enhancing management efficiencies and reducing costs.

1.3.2 Lake ecology

Lake ecosystems comprise living or 'biotic' plants, animals and microorganisms, as well as non-living or 'abiotic' physical and chemical components. The principal abiotic factors in lakes include turbulence from wind, temperature, pH, the availability of light, carbon, nutrients and oxygen, and habitat permanence (Moss, 2007). These abiotic characteristics constitute the abiotic frame within which the biotic organisms live, which is essentially the sum of all the physical and chemical characteristics of a specific lake (Bronmark & Hansson, 2005). The abiotic frame of a lake experiences frequent variations and temporal fluctuations. As a result, biotic adaptations of organisms within lakes show no taxonomic borders as tolerance of low oxygen levels or pH are found across the aquatic food web, from bacteria to plants and fish (Bronmark & Hansson, 2005).

The organisms constituting the biotic component of a lake range in size from minute

viruses and bacteria (0.001 to 0.007mm) to large organisms such as fish, amphibians and birds (Greve, 2003). The majority of aquatic organisms, however, range in size from 0.01 to 1mm and include groups such as protozoa, rotifers, most algae and many macrozooplankton (Moss, 2007). As well as size, freshwater animals are commonly divided into different groups based on their ecology, for example into filter feeders, predators, planktonic and benthic groups (Andersson et al, 1978; Lazzaro, 1987; Richardson & Mackay, 1991; O'Brien, 2002).

Aquatic plants include substrate-attached macrophytes, free-living phytoplankton and periphytic algae, which are also substrate dwelling (Bronmark & Hansson, 2005). Primary producers such as these have the same resource requirements including nutrients (mainly phosphorus and nitrogen), as well as carbon dioxide and light, which they rely on for photosynthesis and nutrition (Jeppensen et al, 1997). The main feature of macrophytes is their green colour as a result of the pigment chlorophyll within their cells that gathers energy from sunlight, although they do possess different solutions and adaptations to optimise resource intake (Bronmark & Hansson, 2005). The primary advantage macrophytes have over their terrestrial counterparts is they do not depend on water supply and their stems and leaves can be supported by the viscosity of water and thereby reduces the need for energy to create growth of support tissue. There are four types of macrophyte classified initially by Sculthrope (1967) and then by Cronk & Fennessy (2001) as emergent, submerged, floating-leaved and free-floating (Figure 1.5).

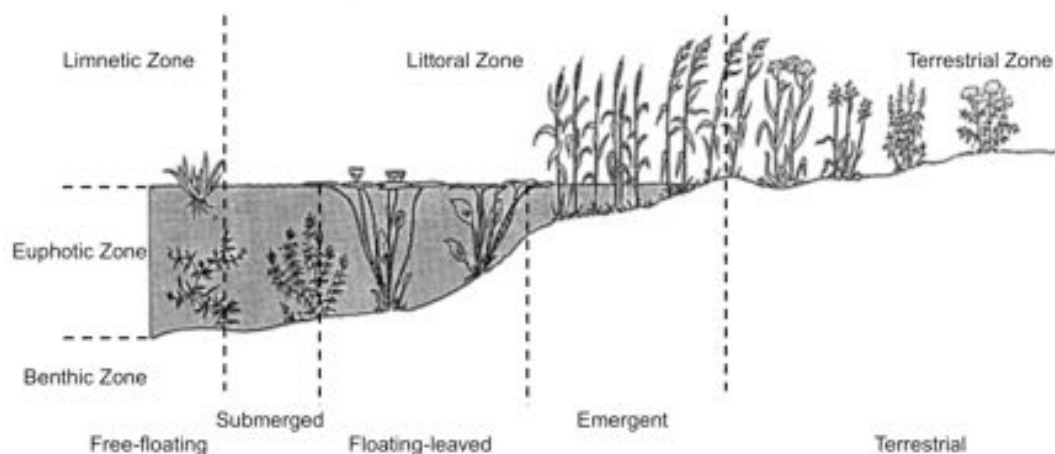


Figure 1.5 Lake macrophyte and ecological zonation including the open water euphotic zone and the benthic zone. Note that the benthic zone representation has been simplified as it actually extends all the way to the emergent zone at the lakeshore.

Emergent macrophytes, anchored to the substrate by their roots, are so named because their basal portions typically grow below the water surface, while their leaves, stems and reproductive organs are largely aerial (Squires & Van der Valk, 1992; Bronmark & Hansson, 2005). Emergent macrophytes gain nutrients from the substrate and photosynthesize above the water surface, and are arguably the most similar to terrestrial

flora (Shipley & Keddy, 1988). Some emergent macrophytes such as *Phragmites australis* form dense monoculture stands in shallow water and spread rapidly thanks to below ground rhizomes from which new shoots grow (Graneli et al, 1992). Emergent species have seeds that can survive for many years in sediment in the form of a seed bank, which they use as a form of insurance for survival during catastrophic events especially when water levels are significantly reduced (Welling et al, 1988). Some of the most common emergent species include those from the *Poaceae* (grasses), *Cyperaceae* (sedges), *Juncaceae* (rushes) and *Typhaceae* (cattail) families.

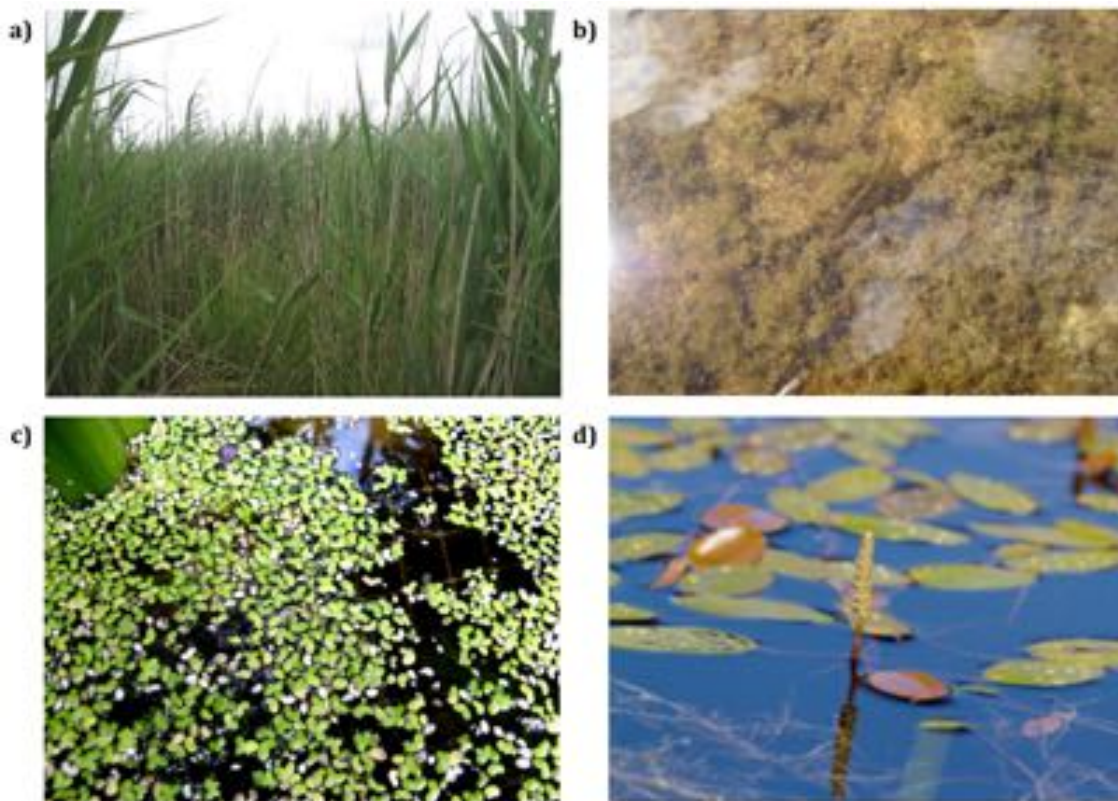


Figure 1.6 Examples of aquatic macrophytes: (a) Emergent (*Phragmites australis*); (b) Submerged (*Chara aspera*); (c) Free-floating (*Lemna minor*); (d) Floating-leaved (*Potamogeton natans*).

Submerged macrophytes typically grow underwater for their whole life cycle, with the exception of flowering periods for some species (Carpenter & Lodge, 1986). These plants attach to lake substrates by roots for the main purpose, unlike terrestrial plants, of anchorage rather than absorption of nutrients and water from the sediment (Barko & James, 1998). Cook (1996) reported that photosynthetic tissues in submerged macrophytes normally exist underwater and that these plants get most of their nutrients through their leaves, which also lack the external protective tissues required by terrestrial plants to limit water loss. Submerged species can be grouped into three key families including *Callitrichaceae* (water starwort), certain *Potamogetonaceae* (pondweeds), and *Hydrocharitaceae* (frogbit) (Singh, 2010).

Free-floating macrophytes are those that possess leaves and stems that primarily grow on the water surface (Meerhoff et al, 2003). If roots are present, they float free in the water column and are not anchored to the substrate. Floating macrophytes therefore move

around on the surface of the water depending on wind and water current (Bronmark & Hansson, 2005). There are several types of free-floating macrophytes and they include *Lemna* (duckweed), the fern-like *Azolla spp.*, *Eichhornia crassipes* (water hyacinth) and *Elodea Canadensis* (Canadian pondweed) (Singh, 2010).

Floating-leaved macrophytes are so called because their leaves float on the surface while their roots are anchored to the substrate (Malthus et al, 1990). Most floating-leaved species have circular, oval or cordate leaves with entire margins and a tough leathery texture that reduces tearing, especially in higher energy aquatic environments (Bronmark & Hansson, 2005). Some species, such as *Ranunculus flabellaris*, however have underwater leaves in addition to floating leaves (Cook, 1969).

Due to their different life forms, aquatic macrophytes generally occur at different depths, often defining distinct zones between shore and deeper water (Chambers & Kalff, 1985). This zonation (Figure 1.5) is present in most lakes and starts with emergent species close to shore, followed by floating-leaved, free-floating and then submerged macrophytes in deeper waters (Engels, 1988). Water level regime is therefore key, and Mitsch and Gosselink (2000) describe lake hydrology as a double-edged sword in term of its effect on species composition and diversity. Hydrology acts as a limit to species richness but is also a stimulus driving productivity. Ultimately the outcome largely depends on the frequency and magnitude of water level fluctuations and also on the degree of natural and human disturbance (Grimm & Backx, 1990; Gulati et al, 1990; Jeppesen et al, 1997). The global distribution of wetland plants, however, depends on the distribution and types of wetland ecosystem. Some wetland species, like *Phragmites australis* and *Eichhornia crassi*, have extensive geographical distributions that range over several continents. Others however, are endemic to a small area or certain wetland types. Macrophytes exert significant controls on lake productivity and biogeochemical cycles as they occupy key interfaces within lake ecosystems, most notably providing the living link between sediments and water column where they can potentially intercept or modify material flows from land to the pelagic zone (Carpenter & Lodge, 1986; Jacobsen et al, 1997).

1.3.3 Lake hydrology

Lakes are a key part of the hydrological cycle (Winter, 2004). Interacting directly with atmospheric water, surface water and groundwater, lake hydrology is greatly influenced by both physiographic setting, which affects movement of surface water and groundwater, and by climatic setting, which affects atmospheric water components, precipitation and evaporation (Winter, 2004). As a result, naturally formed lakes tend to be concentrated in regions where natural terrestrial depressions exist and where climate is predominantly wet (Meybeck, 1995). Water volumes held within and that pass through lake systems are an important component of the hydrological cycle, which explains why many studies on lake hydrology are focussed on lake water volume (Thornthwaite & Holzman, 1939; Thornthwaite & Mather, 1955; Chang, 1965; Piper et al, 1986; Kebede et al, 2006). Variation in lake volume also plays a role in the type and rate of biogeochemical processes,

and many biogeochemical studies are also concerned with volume and the residence time of water in lakes (Kelly et al, 1987; Jefferson Curtis & Schindler, 1997; Schindler, 2006).

Water may flow in several ways into, through and out of a lake (Figure 1.7). The potential sources of water to lakes, which vary greatly depending on physiography and climate, are direct precipitation, inflowing streams or rivers and groundwater (Winter, 2004). Removal of water from lakes occurs through evaporation, outflowing streams and groundwater (Winter, 1981; Winter, 1995). These are the major components that influence the hydrology of most lakes.



Figure 1.7 Flow diagram showing the key hydrological components associated with lake water budget.

Precipitation: Arguably the most dynamic component of the hydrological cycle and lake water budgets is atmospheric water (Winter, 2004). The distribution of precipitation can vary widely in time and space. For example, lakes in the mid-latitudes are often subject to convective storms, which produce highly variable precipitation through the year (Penman, 1948). Establishing precipitation volume input to lakes would ideally require direct measurement at the lake surface. This is often impractical, and the majority of precipitation-based studies use data collected from nearby land-based stations (Lamb, 1966; Nicholson, 1999; Tsintikidis et al, 2002). Calculations based on precipitation gauging often contain uncertainties associated with the instrument of spatial variations (Tsintikidis et al, 2002). Where resources are sufficient, data can be collected from a network of gauges to reduce these uncertainties (Kattelman & Elder, 1991; Winter, 2004).

Evapotranspiration: The combined loss of water through evaporation and plant transpiration is usually estimated using more commonly monitored weather variables as the most accurate methods of measurement are highly resource intensive. A large number of empirical methods have therefore been developed over the last 50 years to estimate

evapotranspiration from climatic variables including solar radiation, wind speed, temperature and humidity. Some of these are derived from the Penman equation, which determines evaporation from grass, otherwise known as evapotranspiration, based on an energy balance and aerodynamic formula (Penman, 1948).

Streamflow: Lakes with inflowing and outflowing streams tend to exhibit water budgets that are dominated by streamflow (Winter, 2004). Streamflow is an element of surface runoff and therefore another primary component of the hydrological cycle but, unlike precipitation and evaporation, it can be measured relatively accurately using gauging stations on inflows and outflows (Winter, 1981; Harmel et al, 2006). Where this is not possible, it is still important to understand the flow regime of inflows, which is largely determined by watershed characteristics (Solokov & Chapman, 1974; Dunn & Mackay, 1995; Hu et al, 2007; Winter, 2004). For example, streamflow will be more erratic in watersheds with steep sides and impermeable substrates, as opposed to watersheds with gentle slopes and significant subsurface storage that will promote more constant streamflow inputs (LeRoy Poff & Ward, 1989).

Groundwater flow: The groundwater-lake interface is the largest and perhaps the most complex boundary of water movement in the hydrological cycle of lakes (Winter, 1981; Krabbenhoft et al, 1990; Wetzel, 2001; Winter, 2004). The rate at which groundwater moves is largely dependent on the permeability and hydraulic properties of the substrate underlying and surrounding each lake, as well as the chemical properties of the lake water itself (LaBaugh, 1988). Even lakes with impermeable substrates will experience a certain amount of percolation and groundwater recharge through bedrock macropores and fractures (Burt & Butcher, 2006; Niedda & Pirastru, 2012). The undulating nature of most lakebeds mean there is a much larger contact area lying between the lake water and substrate than there is between the lake water surface and the atmosphere. Due to this connection, lakes may experience both percolation outflow and seepage inflow from groundwater and the dominance of these processes is largely dependent on the location of the lake (Stauffer, 1985; Winter, 2004). For instance, lakes situated at high elevations relative to the surrounding topography are more likely to contribute to groundwater than those inflow-dominated lakes situated at low elevations sunken amongst the enveloping landscape. According to Winter (2004), the most thorough instrumentation system for determining lakes groundwater interaction is via a network of piezometers, water table wells and equipment to measure seepage across the lakebed. Permanently installed piezometers and wells provide a more continual stream of information on the three-dimensional flow field around the lake; however difficulties in characterising geological boundaries and the distribution of permeability below ground can lead to uncertainties, even with this high-tech infrastructure (Winter, 1999; Winter, 2004).

1.3.4 Lake water level regime

The water level regime of a lake is closely associated with its morphology and is therefore its hydrological signature (Mitsch & Gosselink, 2000; Hollis & Thompson, 1998). Azous & Homer (1997) define hydroperiod as “the seasonal occurrence of flooding and/or soil

saturation, encompassing the depth, frequency, duration, and seasonal pattern of inundation". Lake hydroperiod reflects the balance of inflows and outflows, in conjunction with the critical relationships between lake water depth, area and volume (Hollis & Thompson, 1998). Water-level fluctuations are dominant forces controlling the functioning of all different types of wetland ecosystems (Wilcox & Meeker, 1991; Poff et al, 1997; Singh, 2010). In lakes, the amplitude and frequency of water level fluctuations are important factors influencing both ecosystem function and ecological productivity (Keddy, 2000; Coops & Havens, 2005). Hydrological regime modifies or determines the structure and functioning of lakes by influencing the array of abiotic factors whilst also controlling the composition of biotic communities, especially around lake margins (Hellsten et al, 1996; Baker et al, 2009).

In most lakes, from day to day, there is an imbalance between inflows and outflows, which leads to an increase or reduction in lake water volume (Solokov & Chapman, 1974; Baker et al, 2009). The total effect of this imbalance can be defined by calculating the water balance or water budget of the lake (Equation 1.1), and changes in gains and losses leads to a change in the volume of water stored in a lake (Gasca-Tucker and Acreman, 2000; Bonnet et al, 2008; Baker et al, 2009).

$$\Delta V/\Delta t = P_n + S_i + G_i - ET - S_o - G_o - Abs + Dis$$

where:

$\Delta V/\Delta t$ = change in volume of water storage in wetland per unit time

P_n = precipitation

S_i = surface inflows

G_i = groundwater inflows

ET = evapotranspiration

S_o = surface outflows

G_o = groundwater outflows

Abs = Anthropogenic abstractions

Dis = Anthropogenic discharge inflows

Equation 1.1 Water balance equation; each of the terms can be expressed as depth per unit time (e.g. m month⁻¹, m yr⁻¹) or as volume per unit time (e.g. m month⁻¹, m³ yr⁻¹) (adapted from Mitsch & Gosselink, 2000).

Gauging lake level is therefore needed to establish a stage-volume relationship for a normal range of water level fluctuations. One way to determine this and to predict future changes in lake level is to calculate the water residence time of a lake (Winter, 2004). Put simply, this is the time taken for the water in a lake to be replaced. Residence time can be calculated by dividing lake volume by the rate of outputs i.e. losses via streamflow, evaporation and groundwater percolation (Singh, 2010). The relationship of the three main components of the hydrological system to the residence time of lake water therefore varies greatly between lakes in different hydrogeological and climatic settings.

Permanent changes in lake water level can also occur due to many factors including artificial drainage, damming and climate change. These may then drive significant changes in the growth dynamics of lake flora and fauna, and in extreme cases will result in an ecosystem shift. Water-level variations impart both direct and indirect controls on flora and fauna; for example the water level directly determines the establishment, growth and survival of aquatic macrophytes, and indirectly influences the structure of the aquatic food

web through increased nutrient inputs during wet years and increased grazing during dry years (Van der Valk, 2005). Lake morphometry, especially depth, area, basin shape, and volume control water balance and hydroperiod (Brooks and Hayashi, 2002). Hollis and Thompson (1998) argued that the knowledge of the relationships between water level, area of inundation and volume of water is essential for the understanding of how hydrological inputs and outputs impact ecologically vital elements of a lake including: the level of saturation in the root zone; water depth for diving ducks; and the area of inundation for rice cultivation. This relationship has been employed in many hydrological models (Thompson and Hollis, 1995; Hayashi and van der Kamp, 2000; Brooks and Hayashi, 2002; Gasca-Tucker, 2005) of lakes and wetlands to transform water volumes into ecologically significant depths and areas. As the water level in a lake increases, the total area of surface water also increases resulting in inundation at the peripheries and subsequent total volume increase.

1.4 Lake and wetland ecohydrology

From the preceding review, it is clear that effective scientific assessment and conservation management of lake and wetland ecosystems requires consideration of the inter-relationship between ecological and hydrological components. Plate (1994) was one of the first to understand and emphasize the important influence of biotic factors on the hydrological cycle and hydrology on aquatic biota. The new approach of integrating ecological and hydrological science has been termed 'ecohydrology' or, alternatively, 'hydroecology' (Hannah et al, 2004). These terms have become more prominent in the international scientific literature over the last decade, and although many view the two terms as equivalent (Dunbar and Acreman, 2001; Nuttle, 2002), in practice this concept has not yet been applied extensively as many ecologists tend to use the term ecohydrology (e.g. Zalewski, 2000), and hydrologists use hydroecology (e.g. Dunbar & Acreman, 2001). As there is no specific basis for the use of one term over the other, ecohydrology is the preferred term in this thesis.

Dunbar and Acreman (2001: 1) define ecohydrology as *"the linking of knowledge from hydrological, hydraulic, geomorphological and biological/ecological sciences to predict the response of freshwater biota and ecosystems to variation of abiotic factors over a range of spatial and temporal scales"*. According to Wassen & Grootjans (1996), the discipline of ecohydrology is largely application driven, with the aim of improving understanding of the hydrological factors determining the natural development of lake and wetland ecosystems, especially with regard to ecosystem management and nature conservation (Figure 1.8). In Northern Hemisphere mid- to high latitude lake and wetland systems, the interaction of hydrology and ecological processes influences local stream flow, water quality, and drainage basin geomorphology, and more broadly, the carbon cycle and climate. Any changes in the ecosystem due to factors such as abstraction or pollution can lead to significant changes in regime, structure and functioning, which in turn impacts on floral and faunal diversity. Lowland freshwater lake and wetland systems require sufficient water of adequate quality, at the right time and in the right pattern to maintain a

desired level of ecological health and functioning (Ramsar, 2009). Wassen and Grootjans (1996) stated that ecohydrological research does not usually claim to unravel all ecological and hydrological mechanisms responsible for the observed changes in the species composition of damaged or restored systems, but may contribute to the understanding of how to properly manage these ecosystems.



Figure 1.8 The integration of hydrology and ecosystem science - ecohydrology - in water management (adapted from Nuttle, 2002: 806).

Shallow lakes that lie within extended depressions or floodplains usually comprise a range of ecosystems where the land-water interface is spatially and temporally variable which leads to an extensive zone of periodic wetting, saturation and flooding in response to weather patterns. According to Wetzel (1999) this land-water interface is the most productive region of a lake per unit area. It is dominated by emergent plants that have evolved a number of structural and physiological adaptations to tolerate the hostile anaerobic sediments of lake peripheries, whilst also taking advantage of the abundant nutrient sources across this zone (Table 1.8). Wheeler (1999) observed that the chemical composition of water is a significant factor determining macrophyte distribution and community composition. In addition, numerous studies have examined the relationship between macrophytes and water level (Ertsen et al, 1998; Hardy, 1998; Bartell et al, 1999; Bobba et al, 2000; Reid & Brooks, 2000; Van der Valk, 2005), revealing that the zoned distribution is largely a function of the contrasting water requirements and inundation frequency tolerated by the different macrophytes species.

Table 1.8 The average rates of lake macrophyte primary production. Note the seasonal maximum biomass is approximate and the rates of production vary greatly (adapted from Wetzel, 1999: 279).

Type	Seasonal maximum biomass (g dry m ²)	Average rate and range of production (g dry m ² year ⁻¹)
Emergent macrophytes	2500 (<2 – 9900)	5000 (3000 – 10300)
Temperate	-	3800 (3000 – 4500)
Tropical	-	7500 (6500 – 10300)
Free-floating	(630 – 1500)	(1500 – 4400)
Floating-leaved	140 (25 – 340)	300 (110 – 560)
Submerged	220 (15 – 500)	1300 (100 – 2000)
Temperate	-	600 (100 – 700)
Tropical	-	1700 (1200 – 2000)

Many studies have shown that the distribution, growth and reproduction of lake vegetation in these shallow lake and wetland environments is strongly related to depth, duration and amplitude of the seasonal flooding (Orme, 1999; Mitsch & Gosselink; 2000,

Keddy, 2000), as well as water balance fluctuations and varying water quality (Acreman & McCartney, 2009). Nevertheless, Wheeler (1999) stated that, while it is possible to find broad trends in water level-plant distributions in these systems, the range of conditions occupied by particular species or communities can be wide and may be inconsistent between, or even within, sites. Still, it is clear that water level fluctuations, whether natural or man-made, cause significant shifts in community composition and structure, and those species intolerant of such environmental variations are likely to suffer stress and eventually local extinction (Wheeler, 1999). Conversely, Richter et al (1996) argued that assemblages of macrophyte species might show clearer relationships to water level behaviour than individual species. They claim that effective management of aquatic ecosystems requires existing hydrological regimes to be characterized using biological indicators. Macrophyte communities and other lake-wetland biota would therefore act as natural hydrological indicators to determine the degree of human interference on the ecohydrology of associated ecosystems.

1.5 Coastal freshwater lake-wetland systems

The island nature of lakes means that, to aquatic organisms, they exist and function as 'patches' of suitable habitat in an otherwise inhospitable terrestrial environment (Birge & Juday, 1934). Within many lowland shallow water bodies however, significant and important connectivity is achieved through linkage across adjacent fluvial, wetland and floodplain environments. The presence or absence of fluvial and wetland systems in conjunction with lakes significantly influences the viability and survival of lake species and highlights the importance of obtaining a thorough connected ecological understanding of these systems (Tokeshi, 1994). This is especially important when considering wetlands and shallow lake systems, which are particularly vulnerable to climate change and impacts associated with anthropogenic activities (Clair, 1998; Mortsch, 1998; Kundzewicz et al, 2007; Bates et al, 2008; Thompson et al, 2009).

Freshwater lake and wetland ecosystems are inherently variable in their physical, chemical and biological properties, and as a result this has led to wide-ranging scientific literature documenting the efforts to formalize their key characteristics into effective typologies (Emmons et al, 1999). These work best for conservation management when the classification is simple, and the factors controlling function and behaviour of the system are clearly defined. But in coastal environments, where shallow freshwater lakes occupy wet lowlands, a significant challenge arises due to the presence of multiple ecosystems and the tendency for complex ecohydrological forcing. For example, Sheskinmore Lough lake and wetland system in northwest Ireland, falls into multiple classification categories. Using the Ramsar classification system as a guide, the wetland system can be defined as a non-forested peatland fen with man-made drainage channels as a feature; however it can also be classified as a permanent freshwater marsh on organic soils with emergent vegetation and waterlogged for most of the growing season (Ramsar, 2009). The lake system, in contrast, can loosely be classified as a coastal freshwater sedimentary lake (Karus & Feldmann, 2013). From a conservation management perspective, there is little

guidance gained in the application of multiple classifications to accurately describe a system, and locally focused efforts to understand the ecohydrology of the system are far more informative.

Coastal lakes are generally defined as former gulfs of the sea that contain varying salinity and water volume. They can be partially connected to the sea or completely separated from it through processes of postglacial land uplift or accumulation of sediments transported by wind, water currents or waves (Kjerfve, 1994; Karus & Feldmann, 2013). In the mid-latitudes, coastal lakes have formed in response to Holocene marine transgression (Kjerfve, 1994) that forced onshore sediment transport, sediment infilling and impoundment of water within coastal valleys. Equally, episodes of coastal dune destabilisation and mobility can create sufficient barriers within coastal valleys to force the development of freshwater lakes (Langford, 2003). Coastal freshwater lakes are found on all continents and represent 13% of the coastline worldwide (Karus & Feldmann, 2013). They are most common in northwest Europe, large sections of the southern and eastern coasts of Australia and along the Gulf of Mexico (Marion et al, 1994; Liu & Fearn, 2000; Ticehurst et al, 2007). In Ireland, coastal lakes are commonplace, primarily located along the northwestern, western, southwestern and southern coastlines (Healy, 1998; Good & Butler, 2000; Oliver, 2007). Coastal lakes may also form within the context of deltas where sediment movement and channel dynamics lead to the closure of previous channels (Loffler, 2004; Cieslinski & Drwal, 2005) or deposition by marine processes creates a backbarrier basin (Perez-Ruzafa et al, 2011; Beer & Joyce, 2013). Wind can be an important factor in coastal lake formation, with deflation of sedimentary surfaces and removal of loose, unconsolidated sediments creating depressions that are then filled with water (Washington et al, 2006). This is a common process in coastal dune environments where deflation of hollows or blowouts exposes underlying water tables and facilitates lake formation (Seppala, 1972; Timms, 1986; Timms, 1992).

1.6 Research objectives

The aim of this PhD is to investigate the ecohydrology of the Sheskinmore Lough lake-wetland system on the west coast of County Donegal, northwest Ireland. Although its location on the coast and its sedimentary formation are characteristics typical of a coastal freshwater lagoon, it can also be described as a permanent inland freshwater lake as it is larger than 8 hectares and currently far enough from marine influence to be deemed inland. The lake and wetland contain internationally significant species and habitats and are designated as both a Special Area of Conservation (SAC) and a Special Protection Area (SPA). The system is partly owned by the National Parks and Wildlife Service (NPWS) who are responsible for maintaining favourable conservation status of designated flora and fauna. Due to their concerns over declining water levels, the hydrology of the lake has been manually adjusted over the last decade to raise water levels in order to improve roosting area for overwintering geese, sustain moisture levels for rare species such as *Vertigo geyeri*, and maintain water depths for the endangered *Najas flexilis*.

Scientific knowledge of coastal lakes and wetland systems, their formation, development,

ecology and environmental functioning is limited (Van Groenendael et al, 1993), and in Ireland is negligible (Caffrey et al, 1999). This interdisciplinary study adopts ecological, hydrological and paleolimnological techniques to understand and explain the ecohydrology of shallow coastal lakes and in particular to ascertain the vulnerability of these systems to regional climate change, hydro-geomorphic variation and local anthropogenic practices. The research also aims to contribute to conservation and management strategies for coastal lake and wetland environments, with explicit evaluation in the context of European (Habitats and Birds Directives) and global (Ramsar Convention) policies. In order for conservation of Sheskinmore Lough SPA to be effective in the long-term, it is important that ecological diversity is maintained within a sustainable, naturally functioning system. In order to achieve this, a clear understanding of the inter-relationships across the lake and wetland system must be obtained. The key objectives of this research are:

- 1) To assess the contemporary ecohydrological conditions through examination and analysis of edaphic factors, distribution of vegetation species and assemblages, and hydrological regime.
- 2) To reconstruct the environmental and ecohydrological history of the lake using palaeolimnological techniques to provide a clearer account of the shifts in ecosystem structure and understand climate controls on lake-wetland evolution.
- 3) To effectively model the hydrology and hydrological management of the site in order to understand the behaviour of water levels across the system and their effects on habitat distribution, and to examine changes in these regimes and processes in the context of different management and climate change scenarios.

The research seeks to provide a more complete understanding of the long-term ecology and environment of the site, whilst modelling will facilitate simulation of potential impacts due to future climate change (in the context of IPCC (Intergovernmental Panel on Climate Change) future climate scenarios) and anthropogenic activities. The study will not only improve knowledge of the current ecohydrology within the Sheskinmore system, it will also enhance wider understanding of coastal lake-wetland ecosystems, their formation, development and vulnerability to climate change and anthropogenic pressures. This enhanced understanding will also ensure future conservation management strategies are underpinned by a strong science base to maximise resilience, preserve diversity of habitats and species, and enhance the natural and sustainable functioning of coastal lake and wetland ecosystems.

2 Study area

2.1 Northwest Ireland: ecophysical context

Ireland has over 4000 lakes (loughs), the majority of which are less than 1km² in area (McGarrigle et al, 1990). The largest lake, Lough Neagh, is located in Northern Ireland and is 381km² in area (Table 2.1). Lough Corrib in County Galway on the west coast is the largest lake in the Republic of Ireland at 176km². Despite the presence of such large loughs in Ireland, the majority of lakes are small (<100km² in area) and shallow (<30m depth) (Flanagan & Toner, 1975). A large proportion of the lakes in Ireland are distributed towards the northwest where extensive drumlin swarms are rich in water bodies and upland areas provide the relief necessary for lake formation.

The geology of Ireland comprises a large central lowland region of limestone with a relief of hills surrounded by a discontinuous border of coastal mountains that vary greatly in geological composition and structure (Figure 2.1) (Charlesworth, 1924). The mountain ridges of northwest Ireland are largely composed of granite overlain by blanket peat, fen and raised bog (Charlesworth, 1953; Webster, 2013). It is thought that the highlands of northwest Ireland acted as a centre for a substantial ice cap and, during periods of successive warming and cooling, ice streams and meltwater eroded considerable volumes of material from the landscape (Carter & Wilson, 1993; Burningham, 1999). As a result, several navigable river systems extend inland from the surrounding upland ring (Caffrey et al, 1999). Ireland has seen at least two glaciations and everywhere ice-smoothed rock, mountain lakes, glacial valleys and deposits of glacial sand, gravel and clay mark the passage of the ice (Colhoun, 1971; McCabe et al, 1993). The northwest is extensively covered with glacial deposits of clay, sand and rock, and as a result contains considerable areas of bog and numerous lakes (Charlesworth, 1924; Charlesworth, 1953). The retreat of the glaciers and the unloading of ice from the landscape approximately 16,000 to 22,000 years ago significantly influenced sea levels, both eustatically by raising them and isostatically by land rebound (Carter et al, 1987). It is believed that more recent changes in sea level in northwest Ireland, observed within the last few centuries to decades, are in response to the continued effects of deglaciation and isostatic uplift (Carter, 1982; Burningham, 1999).

Table 0.1 The largest lakes in Ireland, their location and dimensions.

Lake	Area (km ²)	Length (km)	Width (km)	County	Region	Country
Lough Neagh	381	29	18	Antrim, Tyrone, Armagh, Derry	NNE	Northern Ireland (UK)
Lough Corrib	176	43	16	Galway	W	Republic of Ireland
Lough Derg	118	39	13	Tipperary, Galway, Clare	WSW	Republic of Ireland
Lough Erne (Lower)	112	29	19	Fermanagh	NW	Northern Ireland (UK)
Lough Ree	105	25	7	Roscommon, Longford, Westmeath	Central	Republic of Ireland



Figure 0.1 The hydrology and topography of Ireland (adapted from NASA, 2000).

The large lakes of Ireland, including those of the northwest, date back to the Tertiary period and, as their geological history indicates, were formed primarily by glacial processes (Charlesworth, 1953). Located to the east of Donegal Bay, Lower Lough Erne, for example, has a generally flat bed indicating that ice erosion was responsible for its formation (Hardman, 1875; Charlesworth, 1939; Charlesworth, 1953). The deep narrow trench in the bed of Lower Lough Erne, a feature common to other large Irish lakes, confirms that direct ice-erosion along a fault plane was likely responsible (Charlesworth, 1935; Seymour, 1938; Charlesworth, 1953). Charlesworth (1963) claims that the majority of Irish lakes originated by ice action in some form or other, as they are located within the limit of the last glaciation and in an area known to have experienced active ice-erosion. Upper Lough Erne, on the other hand, is located on the soluble Carboniferous limestone geology that spans out from central Ireland and situated amongst glacial deposits called drumlins, nevertheless it displays significant evidence of formation by solution processes (Hull, 1891; Charlesworth, 1953). Charlesworth (1963), however, believes these geochemical processes are secondary to ice-action, recognizing that they have played a more major role in the formation of temporary karst lake systems called turloughs (Charlesworth, 1928; Charlesworth, 1953; Sheehy Skeffington et al, 2006; Williams et al, 2011). The majority of the lakes in northwest Ireland, however, are not situated on soluble limestone, but occur on a mixture of granite, shales and quartzites (Charlesworth, 1963).

Geology significantly influences the hydrology and geochemistry of lakes and wetlands, and this is certainly the case in northwest Ireland. The majority of lakes in this region are exorheic due to the dominant glacial topography, oligotrophic due to the nutrient poor granitic bedrock, dimictic due to their latitudinal location and some can also be acidic due to their location amongst moorland vegetation (Flower et al, 1994; Ragneborn-Tough et al, 1999). The lakes are mainly drainage lakes with inflows and outflows. Most are linked to

upstream lakes and as a result, few are isolated (Webster, 2013). Studies have shown that, although the limnological geochemistry is fairly even across the region, variations do occur (Caffrey et al, 1999; Gibson & Jordan, 2010). Although the majority of the lakes tend to contain low mineral contents due to the dominance of metamorphic granite bedrock, factors such as small-scale geological variation and external natural and anthropogenic influences can complicate the geochemical signal (Flanagan & Toner, 1975; O'Connell et al, 1987; Ragneborn-Tough et al, 1999). Durnesh Lough on the west coast of Donegal, for example, contains extremely high conductivity and elevated alkalinity compared to the majority of lakes in northwest Ireland due to saltwater intrusion impacting the ionic composition of lakewater (Caffrey et al, 1999). The high alkalinity however, is also a reflection of carboniferous limestone, which dominates the bedrock underneath Durnesh Lough (Gibson, 1989).

2.2 The coastal lakes of northwest Ireland

The coastline of northwest Ireland is characterised by rocky inlets, irregular headlands, high cliffs and deep embayments formed from the underlying geologic structure of the Caledonian Orogeny intersected by faults into Pre-Cambrian basement rocks orientated in a northeast-southwest direction (Stephens 1970; Pitcher & Berger, 1972; Long & McConnell, 1999; Burningham, 2008). The highland areas are dissected by numerous glacially deepened valleys formed during the Pleistocene and are surrounded by a narrow fringe of coastal lowlands (Pitcher & Berger, 1972; Shaw & Carter, 1994). The coastline contains several major sea loughs including fjards (glacially-formed marine inlet with shorter, shallower profiles than fjords) such as Lough Foyle and Mulroy Bay in County Donegal, and fiords such as Lough Swilly, also in County Donegal (Figure 2.1) (Pitcher & Berger, 1972). Numerous smaller inlets also line the coastal fringe along with a scattering of small islands. In areas where sediments rather than bedrock dominate, dissipative beaches, large sand-dominated coastal dune systems and shallow, sand-choked estuaries are present (King, 1966; Wilson, 1990; Carter, 1990; Carter & Wilson, 1993; Burningham, 1999). During the late Quaternary, the Devonian granitic rocks of the region were eroded by successive glacial cycles revealing the current coastline geomorphology and lake structures that can be seen today (Colhoun, 1971; McCabe et al, 1993; Knight, 2002; McCabe, 2008). Erosional deposits transported by ice towards the offshore glacial margins of the Atlantic continental shelf dot the landscape, which is underlain by schist, quartzite and granite bedrock (Charlesworth, 1924; Knight et al, 2004).

Prior to the Habitats Directive in 2000, only four coastal lakes in Ireland had received noteworthy research attention: Lady's Island Lake and Tacumshin Lake in County Wexford, southeast Ireland; Lough Murree in County Clare, southwest Ireland; and Furnace Lough in County Mayo, western Ireland (Figure 2.1). This knowledge gap prompted the National Parks and Wildlife Service (NPWS) to commission surveys of coastal lagoons in Ireland (Good & Butler, 1998; Hatch & Healy, 1998; Healy & Oliver, 1998; Oliver & Healy, 1998; Healy, 1999a; Healey, 1999b; Roden, 1999; Good & Butler, 2000). These reports focused on compiling an inventory to facilitate the selection of

protected sites for designation (Healey, 1997). The survey identified 87 coastal lagoon sites, some comprising clusters of very small systems, some saline and others freshwater, covering a total area of 2366.5km² for designation in Ireland making coastal lagoons one of the most completely surveyed habitats in the country. Since this inventory, further research has focused on saline lagoons, but the freshwater coastal lakes still lack attention (Oliver 2005, 2007; Roden 2004). In the original coastal lagoon survey, five main morphological types were identified in Ireland, and it was presumed that this reflected the overall distribution and composition of coastal lagoons (Table 2.2) (Oliver, 2007). The survey found that classic sedimentary lagoons dominate the Irish coastline with 41% of the habitat area, closely followed by artificial lagoons covering 35.2% (Healey, 1997). The survey also revealed that the northwest of Ireland comprises all of the coastal lake types except for karst lakes, which are located in southwest Ireland due to the dominant limestone geology in that coastal region (Table 2.3). Artificial lagoons, however, are the most abundant coastal lagoon type with 30 identified compared to 21 sedimentary lagoons out of the 87 surveyed. Many parts of the Irish coastline especially along the eastern and southeastern coasts were drained in the 18th century (Healey, 1997; Oliver, 2007). In addition, some of these areas may have contained small short-lived lagoon systems that no longer exist due to natural and anthropogenic reasons. Although there are no data available for historical loss of coastal lagoons, it is well known that many artificial coastal lagoons have been created, which may have compensated for some of the losses (Healy & Oliver, 1998; Oliver, 2007).

Table 0.2 The five main morphological coastal lagoon types in Ireland, their location, the number surveyed and their dominance in terms of the coastal lake habitat surveyed (Adapted from Oliver, 2007; p34).

Lagoon Type	Location	Number Surveyed	% Surveyed Habitat Area
Sedimentary	Whole coastline	21	41.4
Artificial	Whole coastline	30	35.2
Rock/Peat	West coast	18	20
Karst	Southwest coast	11	4.5
Saltmarsh	West coast	6	1.5

As well as historical losses, reductions in total coastal lagoon habitat area were recorded during the survey period. During this time, six coastal lagoons were identified as being actively drained. Although no lakes were drained completely, it was estimated that 10.6% of the total coastal lagoon habitat area has been affected (Healy & Oliver, 1998; Oliver, 2007). The survey also highlighted that the most significant threat to coastal lagoon ecosystems is not necessarily the loss of total habitat area, but the deterioration in habitat quality due to pollution.

Table 0.3 Coastal lagoons identified in northwest Ireland during the NPWS survey, their size and type (Adapted from Oliver, 2007).

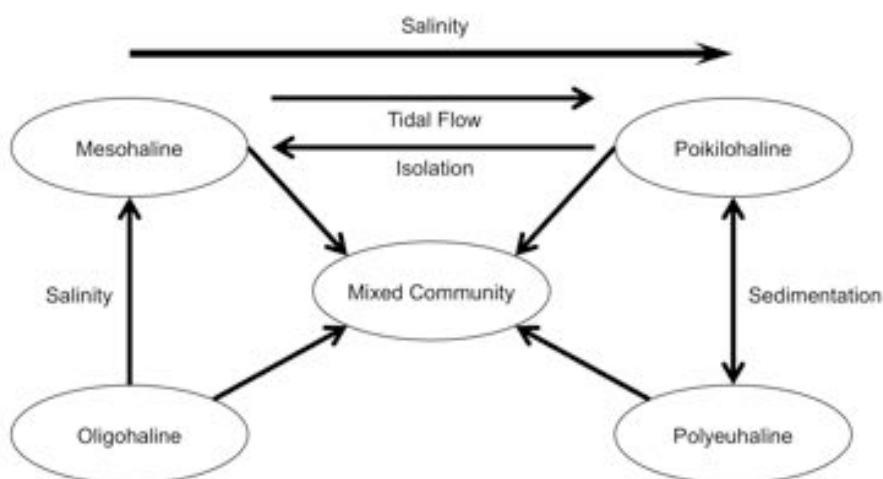
Name	County	Size (km²)	Lagoon Type
Portavaud, Ballysadare Bay	Sligo	6	Saltmarsh
Tanrego	Sligo	2.5	Artificial
Durnesh Lough	Donegal	83	Sedimentary
Maghera Lough	Donegal	19	Rock/Peat
Sally's Lough	Donegal	6	Rock/Peat
Kincas Lough	Donegal	6	Rock/Peat
Moorlagh	Donegal	10	Rock/Peat
Lough O'Dheas, Tory Island	Donegal	3	Sedimentary
Carrick Beg Lough	Donegal	2	Artificial
Blanket Noock Lough	Donegal	40	Artificial
Inch Lough	Donegal	160	Artificial

Coastal lakes are highly dynamic landforms containing transitional ecosystems that are potentially impacted by both marine and fresh water. Environmental conditions within coastal lakes vary considerably as a result (Van Groenendael et al, 1993). Salinity, pH, temperature and turbidity all fluctuate both on a spatial and a temporal basis. Consequently, data collected on an isolated basis from these habitats is likely to be a snapshot in time of biotic and abiotic conditions and less likely to be representative of lake conditions over time (Heegaard, 2004). Coastal lake ecosystems are therefore extremely difficult to study, especially when it comes to measuring ecosystem condition over a short time period. For example, Oliver (2007) believes it is possible that algal blooms observed in some smaller lakes may not reflect their permanent state, but just part of a natural cycle. Nevertheless, they conclude that increases in water pollution due to excessive nutrient inputs from agricultural runoff, urbanisation and industrial sources are unquestionable.

Pollution enhanced by climate change is now considered the most serious threat to the structure and function of coastal lake habitats (Oliver, 2007). Although coastal lagoons retaining some connection to the marine environment contain biota that are characteristically tolerant of extreme variations in environmental conditions and can endure most of the stresses caused by nutrient enrichment, there are limits to such tolerance, especially when considering the large number of lakes that are brackish or freshwater (Table 2.4) (Van Groenendael et al, 1993; Oliver & Healy, 1998; Healy, 2003). Indeed, many freshwater species that are less tolerant of nutrient enrichment can be found in these brackish or 'mesohaline' coastal lakes (Figure 2.2). Algal blooms and fish-kills are therefore becoming more frequent in Ireland and have even been reported from several of the largest coastal lakes (Van Groenendael et al, 1993). In Lady's Island Lake on the southeast coast of Ireland, for example, large areas of aquatic macrophytes and associated fauna have been lost to eutrophication (Healey, 2003).

Table 0.4 The typical species found in coastal lakes according to conductivity (Adapted from Oliver, 2007: p24).

Lake Type	Conductivity	Typical Species
Sedimentary	High to low	Saline, brackish and freshwater specialist species
Artificial	High to low	Saline, brackish and freshwater specialist species
Rock/Peat	High	Salinity specialist species
Karst	High to low	Saline, brackish and freshwater specialist species
Saltmarsh	High	Estuarine 'soft-bottom' species

**Figure 0.2 Conceptual model showing coastal lake community types due to salinity and other factors including tidal influence, isolation from marine processes and sedimentation (Adapted from Oliver, 2007, p25).**

In 2007, more than 80% of the total coastal lake habitat area in Ireland was considered to be 'Unfavourable' in condition (Oliver, 2007). The latest Water Framework Directive status results for Ireland from 2010 to 2012 reveal that 56% of Irish lakes are in moderate to poor condition (EPA, 2016). According to EU guidelines, it requires just 25% of sites to be unfavourable for the whole habitat type to be regarded as 'Unfavourable-Bad' (Bamber, 2010). It is therefore evident that coastal lake biota are under extreme pressure (Service et al, 1996). The benefit of classifying habitat condition has been the rise in conservation management strategies aimed at reducing negative impacts on coastal lakes and restoring them to 'Favourable' status. More work remains to be done to identify anthropogenic pressures and their resultant impacts on those coastal lakes that have, until now, escaped the attention of the early investigations.

2.3 Sheskinmore Lough

Sheskinmore Lough, a shallow coastal freshwater sedimentary lake, is fed by the Duvoge and Abberachrin rivers and is located a few kilometres north of Ardara in County Donegal, northwest Ireland (Figure 2.3). It lies in a backbarrier setting landward of the Magheramore dunes at the mouth of the Loughros More estuary. The system is designated under the EU Birds and Habitats Directives; the Sheskinmore Lough Special Protection

Area (SPA) covers 5.63km² of Natura 2000 Atlantic Biogeographic coastal habitat supporting a plethora of rare and endangered floral and faunal species (NPWS, 2005). The site also lies within the more extensive West of Ardara/Maas Road Special Area of Conservation (SAC) and contains a number of priority habitats listed on Annex 1 of the Habitats Directive including machair and fixed grey dunes. The solid geology underlying the site is almost completely obscured by dune sand except for some exposures of granodiorite fringing the Ardara Pluton to the southwest side of the lake and hinterland to the north and east (Burningham, 1999). To maintain water levels for roosting birds and rare plants within the lake, a sluice (located in Figure 2.3) was installed in the outflowing river by the National Parks and Wildlife Service (NPWS) to actively manage the water levels within the lake (NPWS, 2005). Figure 2.3 also shows the location of the caravan park in the northwest of the SPA.



Figure 0.3 Sheskinmore Lough SPA, its location in County Donegal and Ireland, and the position of the lake, the inflowing Abberachrin and Duvoe rivers and the outflow relative to the Loughros More estuary (Bing, 2012).

2.3.1 Hydrogeomorphological development

Donegal lies within the high-energy northeast Atlantic wind and wave climate region (Rohan, 1986). According to geographical classification of coastal zones, the region is akin to the southwest coasts of South Africa and Chile, and the arctic coasts of Norway, Greenland and Nova Scotia (Davies, 1972). The climate of coastal western Donegal is wet and windy, with an average annual hourly speed of 8ms⁻¹ and prevailing winds primarily from the south to southwest. Much of the west coast of Ireland lies within the tracks of decaying mid-Atlantic hurricanes and extra-tropical storms from the east United States

coast (Cooper & Orford, 1998). During the regular periods of gale force winds that occur during the winter, wind speeds can greatly exceed the annual average of 17.1ms^{-1} . On the whole, this onshore Atlantic wind climate strongly affects the morphology of the northwest Ireland coast (Delaney & Devoy, 1995; Cooper et al, 2004; Burningham, 2008). The region receives 1150mm of precipitation per year and the mean annual temperature is 9.8°C (Knight & Burningham, 2007; Knight & Burningham, 2011). Consequently, the coastal areas and surrounding uplands are largely devoid of woodland, except for scattered softwood forestry plantations dating from the late 19th century (Shaw & Carter, 1994). Rivers on the west coast of Donegal are relatively small, although only the Owenea, one of the larger rivers, is gauged with an average discharge of $7\text{m}^3\text{s}^{-1}$ (1972-2015) just before it enters Loughros More estuary.

Since its formation, the lake and its surroundings have undergone significant changes contributing to the present complex hydrogeology and associated ecosystem (NPWS, 2005). In particular, the ecohydrological regime of the lake and wetland is complicated by the sedimentary drape of calcareous dune sand over acidic peatland and metamorphosed granitic bedrock (NPWS, 2005). The large fixed grey dunes at Sheskinmore Lough are thought to have formed about 5,000 years ago when a slight fall in sea level during the Holocene allowed for the release of large volumes of sub-aqueously stored material to be moved onshore (Shaw & Carter, 1994; Barrett-Mold & Burningham, 2010). About 4,000 years ago the sediment budget significantly declined and, as a result, the dunes have since been reworked and, on a number of occasions, substantially modified (Carter & Wilson, 1993). The lake lies along the shoreline fringe of a glacially-derived drainage basin less than a kilometre from the Loughros More estuary within Loughros Bay (Figure 2.3). This coastal system is typical of most glacially-derived northwestern Ireland estuaries and is dominated by fine quartz sand, with small zones of muddy or gravel sediments (Knight & Burningham, 2007; Knight & Burningham, 2011). Loughros More estuary and the adjacent Loughros Beg estuary have not been significantly influenced by human activities and there are no engineering constraints (Burningham, 1999). The site is underlain by pelite bounded to the east by diorite and granodiorite. The lake is bounded to the east and north by steep rocky escarpments. Small bedrock outcrops protrude through the intertidal flats of the Loughros More and Loughros Beg estuaries that are also bounded by a series of glacial drumlins formed of a matrix of cobbles and silty-sand (Burningham, 1999).

The lake receives fluvial inputs from vast upstream peat bogs that retain significant water volumes in the form of numerous isolated upstream lakes and ponds (King, 1965). The hydrological inputs that arise from these peat systems are minimal, however, in comparison to other hydrological components that dominate the hydrological cycle of the surrounding area, most notably precipitation. Nevertheless, the effects of fluvial inputs on the lake are important. A channel feature, which dissects the lakebed from the northeast to the southwest is clearly visible in recent satellite images (Figure 2.4). Previous studies have suggested that, at some stage in the past, seawater had access to the area that is now dominated by the lake, and the Duvoge valley to the northeast (Shaw & Carter, 1994; Roger Flower pers. comm. 24th October 2011). Indeed, there is also evidence to suggest that the lake and wetland system formed circa 1,000 years ago when sand dune

remobilisation separated the Duvoge and Abberachrin river branches from the main Loughros More estuary, resulting in impoundment of these small rivers (Shaw & Carter, 1994). Today, the lake and SPA lie within a drainage basin that extends into hills up to 60m above sea level; the basin has a mean altitude of 15m above sea level.



Figure 0.4 The channel feature in Sheskinmore Lough highlighted within the 2012 satellite image (Bing, 2012).

2.3.2 Ecology and conservation

The site is of great ecological interest containing at least twenty-three habitats, which are listed on Annex I of the EU Habitats Directive. The site is a candidate SAC due to the presence of blanket bog, machair, fixed grey dunes, decalcified dune heath, decalcified *Empetrum* dunes, and orchid-rich calcareous grassland, which are all priority habitats (NPWS, 2006). The site is also selected as a candidate SAC for habitats listed on Annex I of the E.U. Habitats Directive including: lowland dunes with creeping willow; dune slacks; Marram dunes; large shallow inlets and bays; tidal mudflats; estuaries; Atlantic salt meadows; Mediterranean salt meadows; lowland oligotrophic lakes; alpine heath; dry heath; wet heath; *Molinia* meadows; lowland hay meadows; alkaline fens; Rhynchosporion; and Juniper scrub (NPWS, 2006). Figures 2.5 and 2.6 review the habitats and environments of Sheskinmore Lough and its adjacent wetlands.

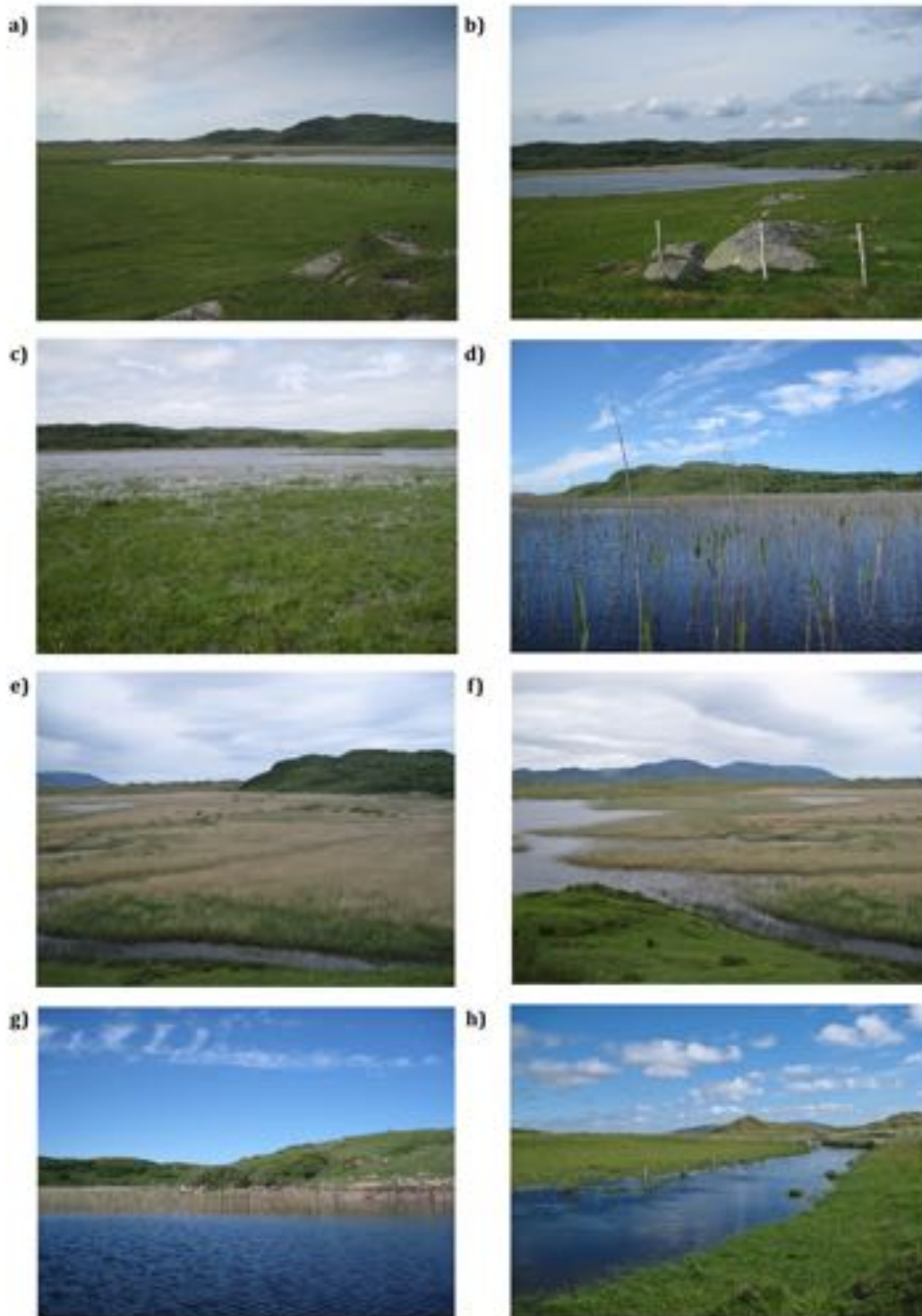


Figure 0.5 Selection of photographs of the Sheskinmore Lough lake system: a) the western half of the lake with reedbed and wetland beyond, viewed from the southwest; b) the northeastern half of the lake with the Duvoge river mouth beyond, viewed from the south; c) the wetland at the southern side of the lake during high water level conditions, viewed from the south; d) the reedbed fringe along the northern side of the lake with uplands beyond, viewed from the southeast; e) the reedbed and Duvoge river, viewed from atop the cliffs on the eastern side of the lake; f) the reedbed, lake and mouth of the Duvoge river, viewed from atop the cliffs on the eastern side of the lake; g) the cliffs on the eastern side of the lake by the Abberachrin river mouth, viewed from the south; and h) the outflow looking downstream towards the estuary via the sluice and dune system, viewed looking south.

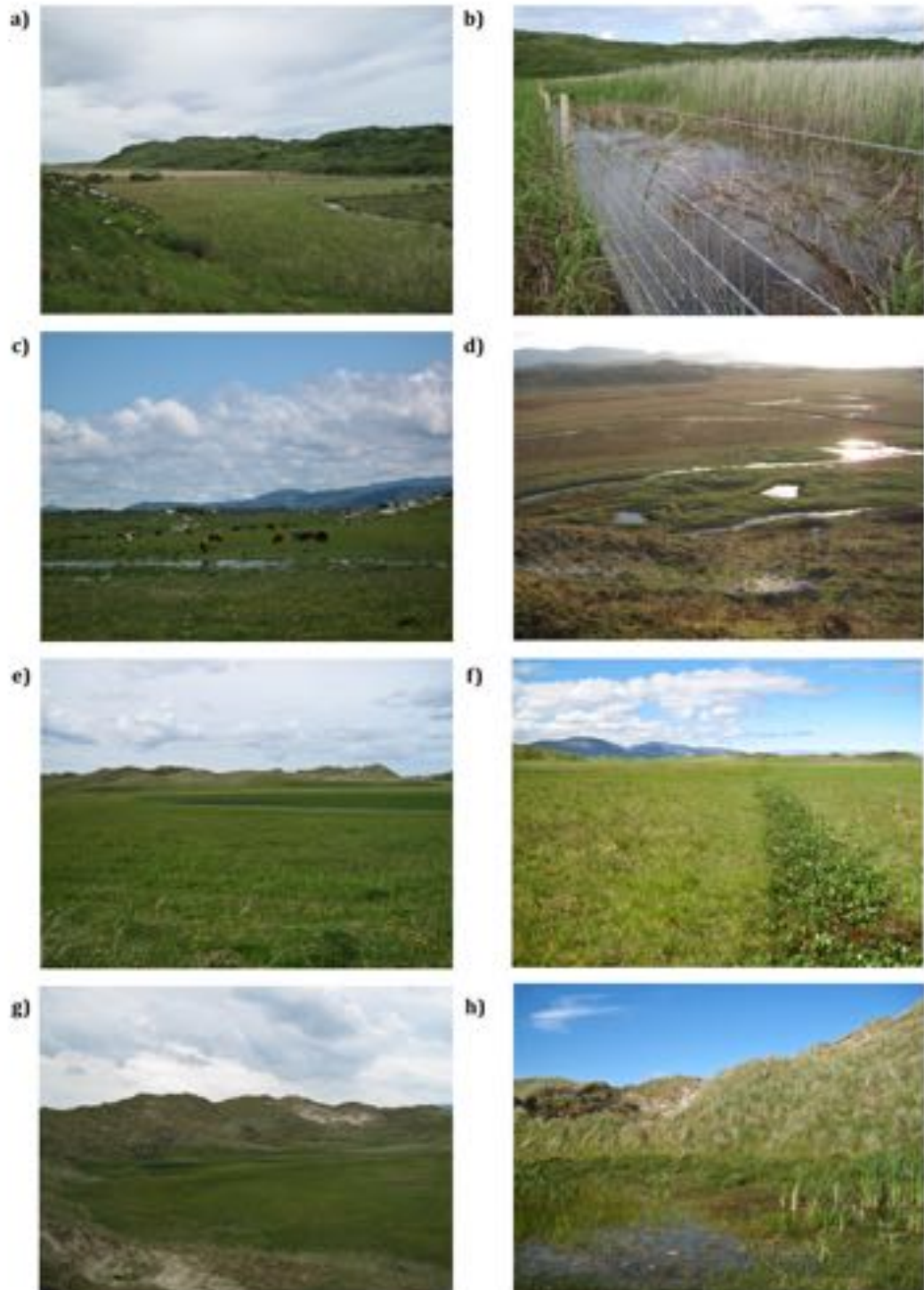


Figure 0.6 Selection of photographs of the Sheskinmore Lough wetland, dune and surrounding systems: a) the Duvoe river floodplain viewed looking downstream towards the lake from the northeast, with reedbed and wetland beyond; b) ponding and the predator fence within the reedbed, viewed looking north; c) cattle grazing the wetland along the southern side of the lake, viewed from the northwest; d) the central and western arm of the wetland system and estuary beyond, viewed from the uplands to the north; e) the western end of the wetland, viewed from the southeast; f) the central wetland system and dunes beyond, viewed looking south; g) the large blowout within the dune system near to the western end of the wetland, viewed looking southeast; and h) one of the ponds on the beach at Tramore Strand at the base of the dune system, viewed looking east.

The site was also selected for internationally important floral and faunal species including: Slender Naiad (*Najas flexilis*), Freshwater Pearl Mussel (*Margaritifera margaritifera*), Marsh Fritillary Butterfly (*Euphydryas aurinia*), Small Blue Butterfly (*Cupido minimus*), Black-tailed Skimmer (*Orthetrum cancellatum*), Petalwort (*Petalophyllum ralfsii*), Atlantic Salmon (*Salmo salar*), Common Seal (*Phoca vitulina*), Geyer's Whorl Snail (*Vertigo geyeri*) and Otter (*Enhydra lutris*), which are all listed on Annex II of the Habitats Directive (NPWS, 2006).

Habitats listed in the SPA designation include sedimentary shallow lake, calcareous grassland, saltmarsh, intertidal sand flats, swamp, fen and wet grassland, which all support a plethora of rare and endangered plant and bird species many of which are protected under EU legislation (Table 2.5). Sheskinmore Lough itself supports a range of charophyte species as well as the rare and protected submerged macrophyte, Slender Naiad (*Najas flexilis*). The lake is surrounded by well developed swamp vegetation dominated by Common Reed (*Phragmites australis*), along with species such as Grey Club Rush (*Scirpus lacustris subsp. tabernaemontani*) and Bottle Sedge (*Carex rostrata*). A small area of fen is found close to the lough containing Black Bog Rush (*Schoenus nigricans*), various sedge species (*Carex spp.*) and Purple Moor Grass (*Molinia caerulea*). Wet grassland is also present close to the lough, with rush species (*Juncus effusus*, *J. acutiflorus* and *J. articulatus*), which dominate, but also a range of herb species such as Cuckooflower (*Cardamine pratensis*), Meadowsweet (*Filipendula ulmaria*) and Tormantil (*Potentilla erecta*) (NPWS, 2005).

Table 0.5 The relative proportions of different habitat areas within Sheskinmore Lough SPA (Adapted from NPWS, 2005: p1).

Habitat Type	Percentage (%)
Inland Water Bodies (Standing Water/Running Water)	5
Dry Grassland	1
Marine Areas (Sea Inlets)	10
Tidal Rivers, Estuaries, Mudflats, Sandflats	20
Saltmarsh, Salt Pasture	5
Coastal Sand Dunes, Sand Beaches, Machair	30
Bogs, Marshes, Fens	20
Heath, Scrub	5
Humid Grassland, Wet Grassland	4

A significant feature of the site is a small area of calcareous grassland with important orchid populations, including the very localised Dense-flowered Orchid (*Neotinea maculata*). The grassland and marsh habitats merge with machair and an extensive sand dune system, which includes Marram dunes, fixed dunes and decalcified dune heath. Marram grass (*Ammophila arenaria*) is frequent along with such species as Sand Sedge (*Carex arenaria*), Red Fescue (*Festuca rubra*), Yarrow (*Achillea millefolium*), Lady's

Bedstraw (*Galium verum*), Bird's-foot Trefoil (*Lotus corniculatus*), Common Dog-violet (*Viola riviniana*) and Pyramidal Orchid (*Anacamptis pyramidalis*). The dune system is bounded to the west by Tramore Strand and to the south by Ballinreavy Strand. Saltmarshes are well developed within the site and typically support plant species including Common Scurvy Grass (*Cochlearia officinalis*), Sea Pink (*Armeria maritima*), Buck's-horn Plantain (*Plantago coronopus*), Sea Plantain (*Plantago maritima*), Sea Milkwort (*Glaux maritima*), Lax-flowered Sea-lavender (*Limonium humile*) and Sea Rush (*Juncus maritimus*). The site also includes the outer part of the Owentocker Estuary where intertidal sand flats are exposed at low tide (NPWS, 2005).

Sheskinmore Lough was formerly of international importance for its EU recognised Greenland White-fronted Goose (*Anser albifrons flavirostris*) and Barnacle Goose (*Branta leucopsis*) wintering populations. Since the 1980s, however, goose numbers at the site have declined markedly and now only a handful of Greenland White-fronted Geese occur during winter. Despite being used regularly as a feeding site during the 1980s and 1990s by a large part (360 individuals) of the western Donegal Barnacle Goose population, Sheskinmore Lough has been abandoned as a feeding site. Nevertheless, this small Greenland White-fronted Goose population is still significant as it is one of the few flocks using traditional habitats. The site also supports small numbers of other waterfowl species including Teal, Mallard, Snipe and Lapwing. Another important feature of the site is the large number of Chough (*Pyrrhocorax pyrrhocorax*) that frequent the area throughout the year, especially in winter. This nationally important population, listed on Annex I of the EU Birds Directive, feed and socialise at Sheskinmore Lough with over 100 birds counted on occasions (NPWS, 2005).

The decline in the populations of wintering geese at this site is not attributable to any changes in habitat quality but rather to a general shift towards improved grassland sites, a trend that has been experienced at many sites in the country in recent decades (NPWS, 2005). A substantial part of the site is owned and managed both by the state-run NPWS and, to a lesser degree by the independent conservation organisation, BirdWatch Ireland, who in turn have management agreements in place with the smaller independent landowners within the SPA. Although the site is infrequently visited and the majority of farming is of low intensity (grazing by a couple of horses in the wetter swamp and fen areas; winter grazing by a small herd of cattle on the broader calcareous and wet grassland habitats), the NPWS have acknowledged that human disturbance, especially from the caravan park could be having a negative affect on bird numbers at Sheskinmore Lough (Table 2.6). It should also be noted that the NPWS believe the management of water levels via the installation of a sluice at the lake's outflow has had a positive effect on lake and wetland biota. This is despite continued depletion in bird numbers and also the rare Slender Naiad (*Najas flexilis*). *Najas flexilis* is one of the main floral assets upon which the designation stands, formally surveyed by Ryan (1981) and Roden (2002). These surveys describe the population as exceeding 1,000 plants, growing in shallow water and scattered in the southern and southeastern parts of the lake amongst *Chara spp.* and under *Sparganium angustifolium* (Roden, 2002; Rostk & Schmidt, 2015).

Table 0.6 The relative proportions and intensities of the various positive and negative anthropogenic impacts on flora and fauna identified within the Sheskinmore Lough SPA (NPWS, 2005).

Anthropogenic Impact	Proportion (%)	Intensity	Influence
Mowing/Cutting	1	Medium	Positive
Fertilisation	5	Low	None
Grazing	50	Low	None
Stock feeding	1	Low	Negative
Camping & Caravans	1	Medium	Negative
Management of water levels	5	Medium	Positive
Erosion	5	High	None

Since its designation, the NPWS (2005) have claimed that the water level in Sheskinmore Lough has fallen in recent years. Although the lake and wetland is classed as freshwater, the underlying acidic bedrock and seepage of basic water from the calcareous dunes presents a challenging and complex hydrogeological regime and associated ecosystem structure. The NPWS have suggested that the water level decline is attributable to natural siltation and ad hoc drainage works (Emer Magee pers. comm. 11th February 2012) but no surveys or monitoring to ascertain aspects of the hydrology, or hydrological requirements of key species and habitats have been undertaken. They stress the importance of maintaining the lake water at a level that ensures a sufficient habitat area for roosting geese is maintained and the optimum depth exists for conservation of Slender Naiad (*Najas flexilis*). Management of water levels to date has largely centred around the sluice, but the paucity of knowledge and understanding of lake evolution, current ecology and related hydrogeomorphic controls, in addition to the natural ecological variability of the system, suggests that continued reactionary hydrological management is unlikely to result in successful and sustainable conservation of this internationally important system.

3 Methodology

3.1 Research design

In order to investigate the past, present and future ecohydrology of Sheskinmore Lough, in relation to regional climate change, hydro-geomorphic variation and local anthropogenic practices, the research methodologies are split into three distinct research strategies: ecology, hydrology and paleolimnology. The research design presented here provides an overview of the approach used to achieve the aims and objectives of the thesis.

Contemporary ecological conditions were assessed and edaphic factors were examined to ascertain the environmental controls on plant species distribution and assemblage. The ecological surveys focus on aquatic macrophytes and were combined with the hydrological monitoring data to determine the effect of hydrological variation on the ecology of the lake and wetland system. The sampling area was defined within a GIS (Geographic Information System) using existing digital maps delineating the main wetland and lake habitat areas within Sheskinmore Lough SPA and discrete sample points were identified using a stratified random sampling approach to ensure even coverage.

The hydrological regime of the lake and wetland was monitored in order to understand the behaviour of water across and within the system, and to examine the hydrological forcing on the ecology. Hydrological monitoring involved logging water levels in the lake and across transects of dipwells extending into the surrounding wetland and dunes. Water level recorders were coupled with flow gauging to establish rating curves that will permit the establishment of long-term records of channel discharge into and out of the lake. An automatic weather station was installed locally within the site to provide estimates of precipitation and evaporation onto and from the lake surface and associated catchment. A detailed bathymetry of the lake and topography of the surrounding dunes was also established using dGPS to provide the morphological context of the immediate catchment. Hydrological monitoring data were used to parameterise, calibrate and validate hydrological models of the lake and wetland system using MIKE SHE, a fully distributed and physically based modelling system (Refsgaard et al, 2010). Calibration and validation were based on a multi-objective approach using observations of levels in the inflowing and outflowing rivers, within the loughs and the elevation of the water table (Thompson, 2004).

Paleolimnological techniques were used to reconstruct the environmental and ecological history of the lake. For conservation of Sheskinmore Lough SPA to be effective in the long-term, it is important that diversity is maintained while also allowing a degree of natural variation to occur. In order to achieve this, a better understanding must be obtained of how far removed the lake and wetland system is from its natural, pre-anthropogenic state. This was achieved by reconstructing the environmental and ecological history of the lake and wetland, using high-resolution paleolimnological, lithostratigraphic and geochemical techniques on cores acquired across the lake and wetland system (Devoy et al, 1996). Diatoms, macrofossils, and sedimentology of accumulated organic and sediment material,

are used here to provide a record of environmental and ecological evolution of this freshwater lake and wetland within the context of the coastal system.

During the field surveys, on-site observations were made and information was gathered from interviews with National Parks and Wildlife Service (NPWS) personnel in relation to land use management and conservation. Research included information regarding hydrological management, agriculture, grazing, mowing, peat extraction and general conservation practices including predator control. Information on the recent history of the site was also gathered from informal interviews from local landowners.

3.2 Ecology

Emergent macrophytes located around the periphery of the lake, in the wetland and reedbed to the west of the lake were surveyed in June 2012, June 2013 and July 2014 to account for inter-annual variability (Figure 3.1). Relationships between vegetation and environmental data were determined using correlation, regression and ordination techniques. These methods include inferential parametric statistical techniques that assume samples are independent of one another and therefore require the adoption of a random strategy as part of the overall sampling method. The wetland area was therefore divided into a numbered grid, with each cell representing a possible sample location. Random numbers were generated, each one corresponding to a different cell within the wetland area. Sample sites were allocated on this random basis and a systematic approach was adopted, especially around the narrow lake fringe areas, to ensure an even coverage of quadrats was achieved. The random element of the method helped minimise bias as every cell had an equal chance of being chosen and the systematic part also ensured a significant proportion of the wetland habitat was sampled (Elzinga et al, 1998). A more systematic approach was adopted in the reedbed habitat. The total area of the reedbed that could be surveyed was limited to the western side due to the erection of a predator fence during the spring prior to the July 2014 survey preventing access to much of the habitat. As a result of the limited access and navigation difficulties associated with sampling in such tall vegetation, sample locations were randomly allocated along a series of evenly spaced transects leading eastwards into the reedbed from the western side.

The standard quadrat method was used to sample the vegetation at each site within the wetland and reedbed. As variation related to the diversity, density, distribution and abundance of plant communities within the two habitats, it was important to choose a quadrat size that encapsulated a portion of vegetation most representative of the overall habitat area. A range of quadrat sizes (1m², 2m², 3m², 4m² and 5m²) were tested within each habitat using a species-area curve as recommended by the Braun-Blanquet Zurich-Montpellier School of Phytosociology as the standard technique for vegetation classification (Kent & Coker, 1992). The relative abundance of species did not increase with quadrats larger than 1m² and so this was chosen as the most appropriate size. This quadrat size also allows comparison with other works on similar habitats, for example the National Vegetation Classification (Rodwell, 2000). The percentage cover of the ground occupied by the vertical projection of aerial parts of each plant species (based on the

National Vegetation Classification System) was measured subjectively within each quadrat by estimations made through ‘visual analysis’ (Kent & Coker, 1992; Rodwell, 2000). Stratification or multiple layering of vegetation often occurred, and so some percentage cover totals exceeded 100%. Species were identified using Haslam et al (1975), Phillips (1980), Jeremy et al (1982), Fitter et al (1984) and Rose (2006). The percentage cover of each species within each quadrat was recorded, as was the proportion of bare ground. Those plants that could not be identified in the field were sampled and identified later; mosses and lichens were recorded, but not identified to species level.

Macrophytes (submerged, floating-leaved and free-floating) located within the main body of the lake were surveyed from a boat in June 2012, June 2013 and July 2014, the former two surveys to account for inter-annual variability (Figure 3.2). It should be noted, however, that a number of the June 2013 sample points were reached by wading where water levels were too low to be accessed by boat. The Percent Volume Infested (%PVI) method, which measures the density of plants underwater at each point, was used to determine macrophyte coverage (Canfield et al, 1984; Zhao et al, 2006). PVI is calculated using the following equation:

$$PVI = \frac{CM}{D}$$

Equation 3.1

where C is the percent coverage of macrophytes, M is the mean height of macrophytes and D is the water depth.

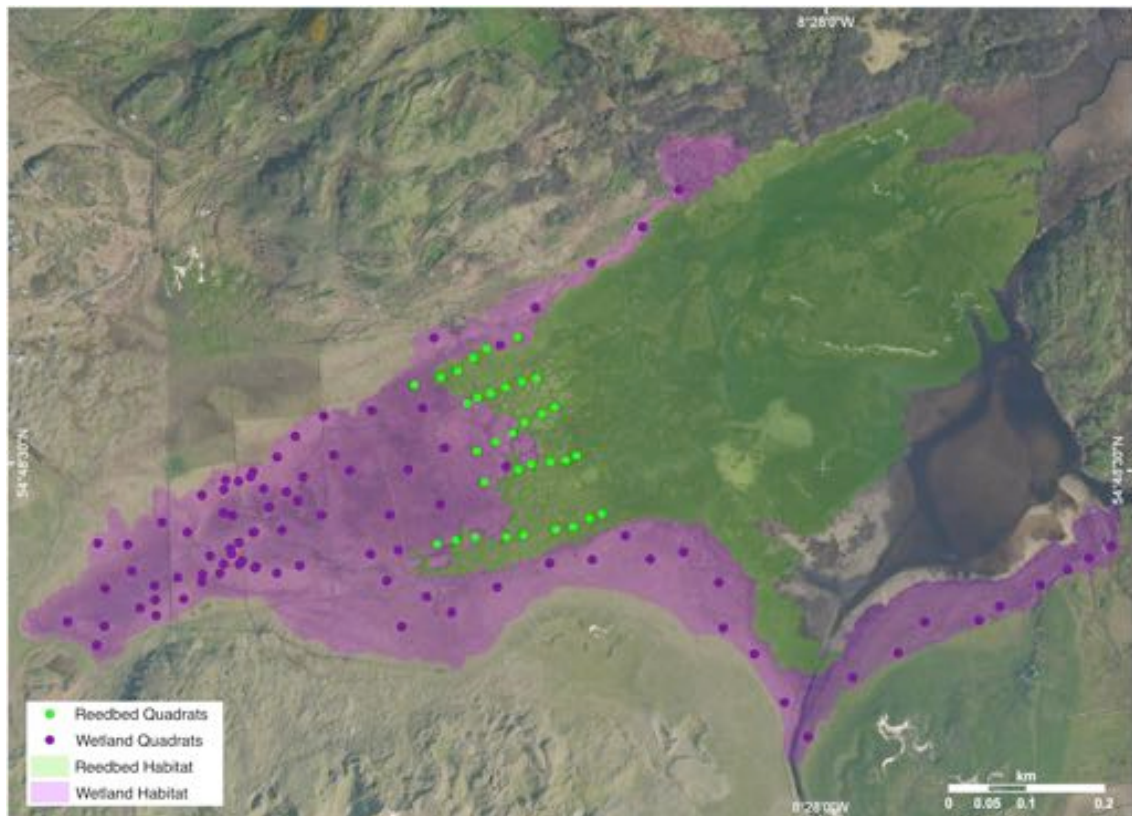


Figure 3.1 Location of the 90 wetland quadrat sites surveyed in June 2012 and June 2013 and also the 33 reedbed quadrat sites surveyed along transects in July 2014 in relation to the total habitat areas. Note that, although some quadrats lie outside the reedbed area delineated from the 2012 satellite image, they are all located within the 2014 reedbed area.

Lake surveys in June 2012 and June 2013 adopted a semi-stratified random approach to locate sample position where sampling was specifically undertaken, both in deeper areas of the lake and along the margins, to ensure an even coverage of samples was achieved. Like the wetland, the open water habitat of the lake was divided into a numbered grid, with each cell representing a possible sample location. Random numbers were generated, each one corresponding to a different cell within the lake area. The random element of the method helped minimise bias as every cell had an equal chance of being chosen and the stratified aspect also ensured a significant proportion of the open water habitat was sampled (Elzinga et al, 1998). In July 2014, however, a more strategic sampling method was adopted to capture the more subtle variations in distribution of macrophyte assemblages across the topographical undulations of the channel feature and across the littoral areas of the lakebed. Sample locations were distributed at regular intervals, every 1m along the north and south channel feature transects, and the northern littoral transect, and every 2m along the southern littoral transect (Figure 3.2). Transect locations were chosen on the basis that they would capture the average, yet potentially contrasting ecologies associated with the different major topographic features and gradients within the lake. Macrophytes were identified using a bathyscope and plant coverage was estimated within the field of view visible through the bathyscope from a stationary point. Water depth and average plant height were measured using a combination of wading rod, secchi disk and depth sounder readings. Species that required closer inspection were brought to the surface using a rake. Charophyte specimens that could not be confidently identified in the field were preserved in Industrial Methylated Spirit (IMS) and identified in November 2012 by charophyte expert and National Stoneworts Recorder, Nick Stewart.

The lake macrophyte communities were also surveyed in August 2013 to locate the rare *Najas flexilis*. A licence to take samples of protected flora from the lake was obtained from the National Parks and Wildlife Service (NPWS) prior to the survey. The search method involved snorkelling following the deeper areas of the lake and manual collection of a single plant sample. GPS points were also taken to record the location of plants. Snorkelling was chosen as the optimum method as the streamline morphology of this plant means recovering identifiable samples would have been extremely challenging using conventional raking procedures and detrimental to the plant itself.

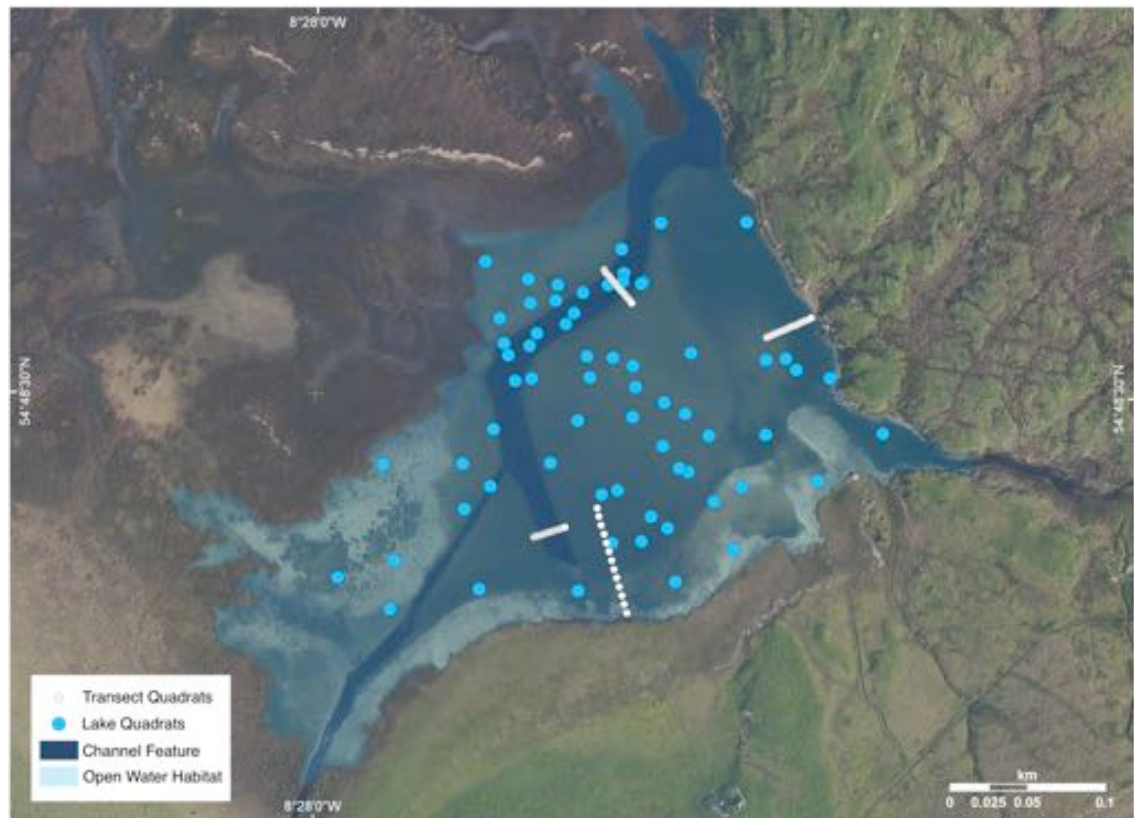


Figure 3.2 Location of the 65 lake sample sites surveyed in June 2012 and June 2013, in relation to the total open water habitat area and the 4 transects located across the highlighted channel feature and the littoral zones surveyed in July 2014.

3.3 Hydrology

3.3.1 Surface and groundwater hydrological monitoring

In June 2012 a comprehensive hydrological monitoring network was installed across the site to monitor the changes in water level (vertical position) within and around the lake and wetland system (Figure 3.3). Sites were chosen on the basis that they captured the dominant channel flow into and out of the lake, and the main pattern of groundwater movement across the wetland and surrounding areas, as well as: 1) accessibility to ensure regular download and monitoring was feasible; 2) water depth to ensure TROLL sensors remained submerged throughout the entire monitoring period; and 3) location to ensure an even spread across the site.

Surface water monitoring sites included Sheskinmore Lough (L3), nearby Sandfield Lough (L6), the two inflows (Duvoge (L1) and Abberachrin (L2) rivers) and outflow (at the sluice gate structure (L4) shown in Figure 3.4, and downstream of the sluice (L5)), two dune-traversing ponds (S1 and S2) and one pond at the back of the Tramore Strand beach (D2). Groundwater monitoring sites included seven dipwells situated along the length of the wetland (W1-4), around the lake fringe (W5-7), one in a large dune slack (D1), one in an area to the north west of the wetland (S3), and two between Sheskinmore Lough and Sandfield Lough (S4 and S5). Hydrological data were collected using Rugged TROLL 100

automatic loggers. TROLLs were deployed underground within constructed dipwells to capture groundwater movement and were also suspended within permanent water bodies to measure surface water level fluctuations. The position of each TROLL was checked during every field visit with hand measurements of groundwater elevation and the logger apparatus to ensure consistency of results.



Figure 3.3 The main hydrological features and the location of hydrological monitoring sites at Sheskinmore Lough. Sites monitoring the lakes and river systems (L1-L6) are shown in blue; sites monitoring the wetland (W1-W7) are shown in red; sites monitoring the surrounding areas (S1-S5) are shown in orange; sites monitoring the dune system (D1 & D2) are shown in yellow; and the weather station (WSt) is shown in purple. Circular symbols indicate dipwell groundwater monitoring sites; square symbols indicate surface water monitoring sites; and the pentagon symbol indicates the weather station.



Figure 3.4 The sluice gate and pedestrian bridge structure observed from two angles at low flow levels in July 2013. The sluice gate comprises a movable gate driven by a hand-operated rack-and-pinion drive mechanism that allows water to flow under it when the sluice is raised. At these times the sluice is 'open'. At times of very high flow when the sluice is 'closed' and no water can pass underneath, water may spill over the top and at these times the sluice gate operates as a weir. Note the rectangular concrete structure with metal grill on the top to the right of the sluice gate houses the fish pass.

Dipwells were constructed from 40mm (\emptyset diameter) white PVC pipes covered by geotextile membrane to prevent blockage by fine silts (Figure 3.5). The diameter was chosen so that it was small enough to ease installation by hand and responsive to actual changes in the water table in the surrounding soil material, but large enough to accommodate the TROLL loggers. The membrane was sealed around the pipes to stop sediment passing into the dipwell via the series of 5mm holes, which were drilled into the sides to allow for free movement of water through the dipwell. Dipwells ranged in depth from 1m to 1.5m due to the presence of underlying gravel and other more solid geologies which proved difficult to penetrate with a hand auger. Dipwells were inserted into holes created using the hand soil auger. At all dipwell locations a larger diameter tube was inserted into the augered hole prior to installation of the smaller diameter dipwell tube in order to stabilise softer layers of sediment prior to dipwell insertion. A TROLL sensor was then suspended on wire attached to the top of the dipwell, which was then capped and taped to prevent rainwater intrusion. The white tops of the dipwells were left standing proud approximately 30cm above the ground surface to ensure they were clearly visible to both humans and livestock. Sediment augered from the dipwells was retained in sequence to allow for simple visual and textural characterisation of the key lithostratigraphical features.

TROLLs located within surface water bodies including the inflows (L1 & L2), lakes (L3 & L6) and outflow (L5) were suspended within 60mm (\emptyset) short lengths of grey PVC tube, which were secured to concrete blocks to ensure the loggers remained stable and upright during periods of high or rapid flow (Figure 3.6). TROLLs located in permanent water bodies, including in the dune ponds (S1, S2 & D2) and at the sluice (L4), were suspended within 40mm (\emptyset) white PVC tubing. In the dune ponds, short lengths (approximately 30cm) of the white PVC tubing were fixed to wooden posts so that the loggers were protected and suspended above the ground surface but below the water surface (Figure 3.7). At the sluice, the TROLL was suspended within a 2m length of 40mm (\emptyset) white PVC tube secured to the existing sluice structure to ensure the logger remained submerged even during low flow periods and when the sluice was open (Figure 3.8).



Figure 3.5 Diagram a) and photograph b) showing the groundwater logger apparatus (dipwell and suspended water logger) and the downloading of a TROLL logger.

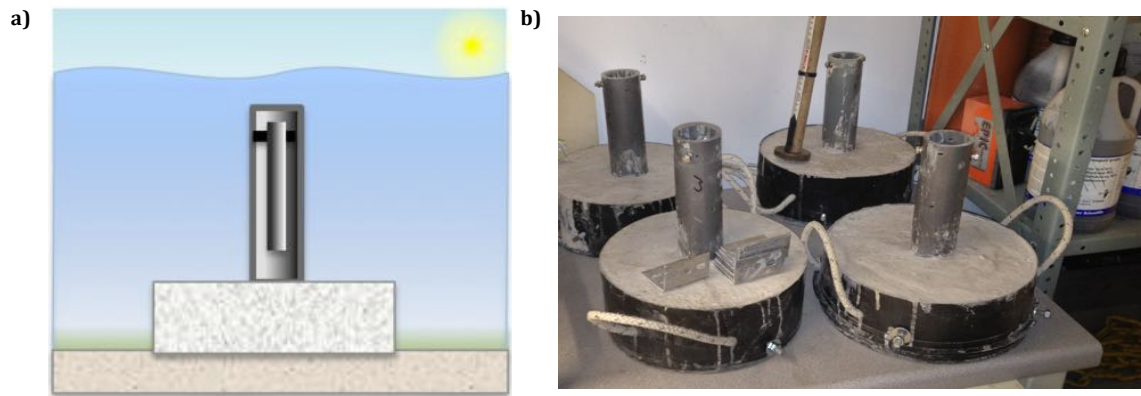


Figure 3.6 Diagram a) and photograph b) showing the surface water logger apparatus (concrete and tube) in the lake and inflows.

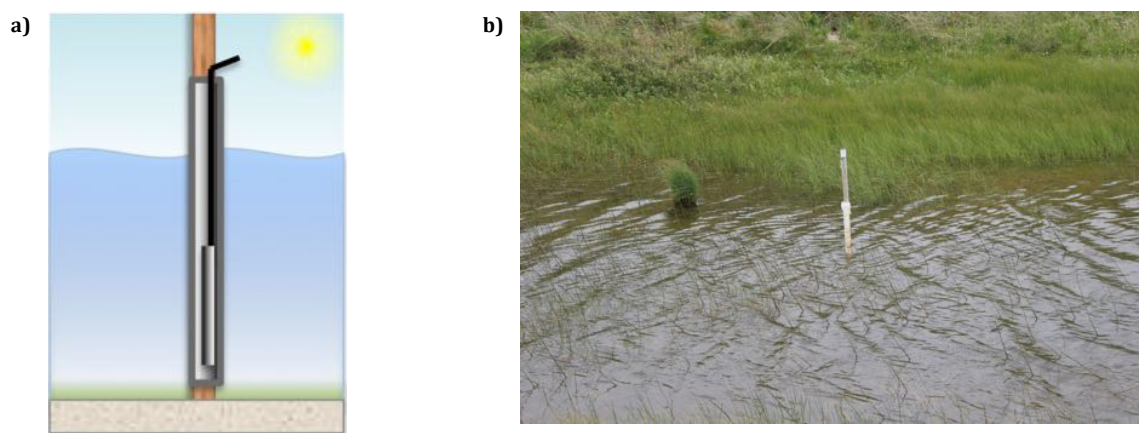


Figure 3.7 Diagram a) and photograph b) showing the surface water logger apparatus (wooden stake and tube) in the dune ponds.

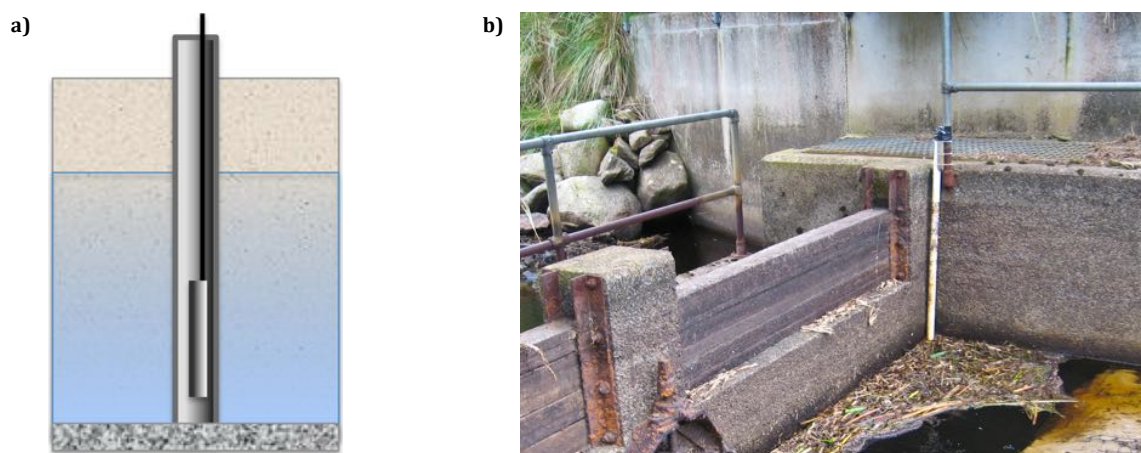


Figure 3.8 Diagram a) and photograph b) showing the surface water logger apparatus (tube and suspended logger) attached to the sluice gate structure.

The TROLLs recorded water depth (derived from pressure head above sensor) at 15-minute intervals, a frequency deemed sufficient for the duration of the monitoring period. Due to their finite data storage capacity, the majority of the TROLLs were downloaded on each visit and a note was taken of the date and time of each download. During a one-week

period in June 2012, an additional CTD (conductivity-temperature-depth) diver was installed in the outflow downstream of the sluice (L5). Secured within a similar tube and concrete block structure to those housing TROLLs in the inflows and lakes, the aim with this logger was to measure water depth and conductivity levels to ascertain the degree of estuarine (tide and salinity) influence on the outflow.

The loggers were programmed to account for freshwater densities prior to deployment. Data were processed using standard pressure-depth conversion calculations. The pressure data were converted to water depth (h , measured in metres) using:

$$h = \frac{P_o - P_a}{100g\rho_w}$$

Equation 3.2

where P_o is logged pressure (mbar), P_a is atmospheric pressure (mbar), g is acceleration due to gravity ($9.81 \text{ m}\cdot\text{s}^{-2}$) and ρ_w is (fresh) water density ($1,000 \text{ kg}\cdot\text{m}^{-3}$). The depth calculations were then converted to water level (distance between water and ground surface) using manual measurements obtained during instrument deployment.

Variable availability of the hydrological monitoring equipment necessitated some degree of adjustment in the deployment location and period, and recording intervals of specific measurements. A year-long survey from June 2012 to June 2013 (365 days 18th June 2012 to 17th June 2013) was conducted for analyses focusing on the long-term behaviour of water levels and their effects on the ecology of the lake and wetland system. A subsequent year-long monitoring survey was also conducted from 18th June 2013 to 17th June 2014 to provide a means for assessment of interannual hydrological variation and also to aid calibration and validation during hydrological modelling. Hydrological monitoring involved logging water levels in the lake, inflows and outflow of Sheskinmore Lough itself (L1-L4), across transects of dipwells in the wetland (W1-W3; W5-W7) and extending into the surrounding dunes (D1 and D2) (Figures 3.9 and 3.10). A TROLL was also installed in the smaller Sandfield Lough in the southeast part of the site.



Figure 3.9 Photograph of the sealed top of a dipwell within the wetland.

Discrete monitoring periods from 7th September to 22nd October 2012 (46 days), from 1st July to 31st July 2013 (31 days) and from 18th to 24th June 2012 (7 days) were undertaken to gauge the hydrology of peripheral sites, and their interaction with the core lake and wetland system, and also to monitor conductivity levels in the outflow to determine the degree of saline influence from the Loughros More estuary. In the June 2012 week-long monitoring period, five additional TROLLs were installed in two dipwells across the land between Sheskinmore Lough and Sandfield Lough (S4 and S5), as well as two TROLLs in two dune ponds at the western end of the site (S1 and S2) and one wetland dipwell on the western fringe of the lake (W4). W4 was put in place close to the existing W3 logger to identify any potential small-scale variations in groundwater level that may occur across the site. A CTD diver was installed at L5 in the outflow downstream of the sluice to assess conductivity as well as water level. In September and October 2012, three TROLLs were reinstated for a 6-week period in their former sites in the two dune ponds (S1 and S2) and wetland lake fringe (W4). In June 2013, the two TROLLs in the dune ponds (S1 and S2) were reinstated for a third time where they remained for the remainder of the monitoring period. It should be noted that TROLLs were never returned to the wetland dipwell at the lake fringe (W4) as this site exhibited little variation from nearby dipwell W3, or the two sites between Sheskinmore Lough and Sandfield Lough (S4 and S5) as sometime between August and October 2012 the dipwells were destroyed by farm machinery and the site was therefore considered too risky. In addition to these temporary monitoring sites, a diver was installed on 8th March 2014 upstream in the Duvoge river at L7. This additional diver was installed as preliminary analyses of the hydrological monitoring data revealed that water level fluctuations at L1 in the Duvoge matched the magnitude and frequency of water level fluctuations within the lake at L3. As this is likely due to the location of L1 in a wide saturated floodplain and the close proximity of L1 and L3 and their similar elevations, the additional diver at L7 was installed. L7 monitored water levels from 8th March to 12th July 2014 to establish Duvoge river inflow levels upstream of any potential hydrological influence from the lake. This additional information also provided a means for conversion of discharge and flow measurements to ensure they represented the catchment as a whole.



Figure 3.10 Location of the hydrological monitoring sites for the two year-long surveys (all coloured symbols except L7) and also for the discrete monitoring periods (all symbols). Sites monitoring the lakes and river systems (L1-L7) are shown in blue; sites monitoring the wetland (W1-W7) are shown in red; sites monitoring the surrounding areas (S1-S5) are shown in orange; sites monitoring the dune system (D1-D2) are shown in yellow; and the weather station (WSt) is shown in purple. Circular symbols indicate dipwell groundwater monitoring sites; square symbols indicate surface water monitoring sites; and the pentagon symbol indicates the weather station. Sites not included in the year-long survey (L5, W4, S1, S2, S4 & S5) are shown as colourless.

3.3.2 Meteorological survey

Meteorological data at Sheskinmore Lough SPA have been collected since 18th June 2012 using a Davis Weatherlink VantagePro2 automatic weather station (AWS) positioned approximately 750m to the southeast of the lake (Figure 3.10 and Figure 3.11). The weather station incorporates a tipping-bucket rain collector that measures 0.25mm for each tip of the bucket, combined with the console, which logs precipitation data and converts it into totals at the time it is displayed. Converting at display time reduces possible compounded rounding errors over time. In addition to measuring precipitation, the weather station houses an outside air temperature sensor in a vented and shielded enclosure. This minimises solar radiation induced error when temperature data are recorded.

During the first year of installation (to 17th June 2013), logging of meteorological data was set to a 30-minute interval, and thereafter (to 17th June 2014) to a 60-minute interval. The full suite of meteorological metrics recorded are provided in Table 3.1; however this study utilised precipitation, air temperature, evapotranspiration, wind speed and wind direction data. Evapotranspiration is a measurement of the amount of water vapour returned to the air in a given area. It combines the amount of water vapour returned through evaporation

(from wet surfaces) with the amount of water vapour returned through transpiration (exhaling of moisture through plant stomata) to arrive at a total. Evapotranspiration is estimated by the weather station by applying the Modified Penman Equation to air temperature, relative humidity, average wind speed, and solar radiation data, and is calculated once an hour on the hour. Wind speed and direction were measured as 10-minute average wind speed and 10-minute dominant wind direction for each archive interval. All weather data were converted to 15-minute intervals by interpolating between readings to allow for direct comparison with hydrological data.

To set the meteorological monitoring period in context of longer-term climate change, historical temperature and precipitation data were obtained from the European Climate Assessment & Dataset (ECA&D) via the Met Éireann website: the Irish National Meteorological Service, a division of the Department of the Environment, Community and Local Government and the leading provider of weather information and related services for Ireland (ECAD, 2015; Met Éireann, 2015). The ECA&D dataset contains series of daily observations at meteorological stations throughout Europe and the Mediterranean. Some of the datasets provided are freely available for non-commercial research and education. The data downloaded were recorded at the Malin Head meteorological station in northern Donegal and span a 58-year period from 1st January 1957 to 31st December 2014. This meteorological station was chosen as the most appropriate out of those available, as it provided the longest and most complete historical dataset, the most similar weather patterns and, like the Sheskinmore Lough weather station, is located within County Donegal in an Atlantic location on the northwest coast of Ireland.

Table 3.1 The full suite of meteorological metrics available with the Davis VantagePro2 automatic weather station (AWS).

Weather Variable	Units
Wind Speed and Direction	miles per hour (mh ⁻¹); kilometres per hour (kmh ⁻¹); metres per second (ms ⁻¹)
Outside and Inside Temperature	degrees Fahrenheit (°F); degrees Celsius (°C)
Humidity	percentage (%)
Wind Chill	degrees Fahrenheit (°F); degrees Celsius (°C)
Dew Point	degrees Fahrenheit (°F); degrees Celsius (°C)
Barometric Pressure	inches (in); millimetres (mm); milibars; hectoPascals (hPa)
UV (Ultraviolet Radiation)	milliwatts per square centimeter (mWcm ⁻²)
Heat Index	n/a
Temperature Humidity Sun Wind (THSW)	degrees Fahrenheit (°F); degrees Celsius (°C)
Rain Rate	inches per hour (inhr ⁻¹); millimetres per hour (mmhr ⁻¹)
Precipitation	inches (in); millimetres (mm)
Daily Rain	inches (in); millimetres (mm)
Rain Storm	inches (in); millimetres (mm)
Solar Radiation	Watts per square metre (Wm ⁻²)
Evapotranspiration (ET)	inches (in); millimetres (mm)

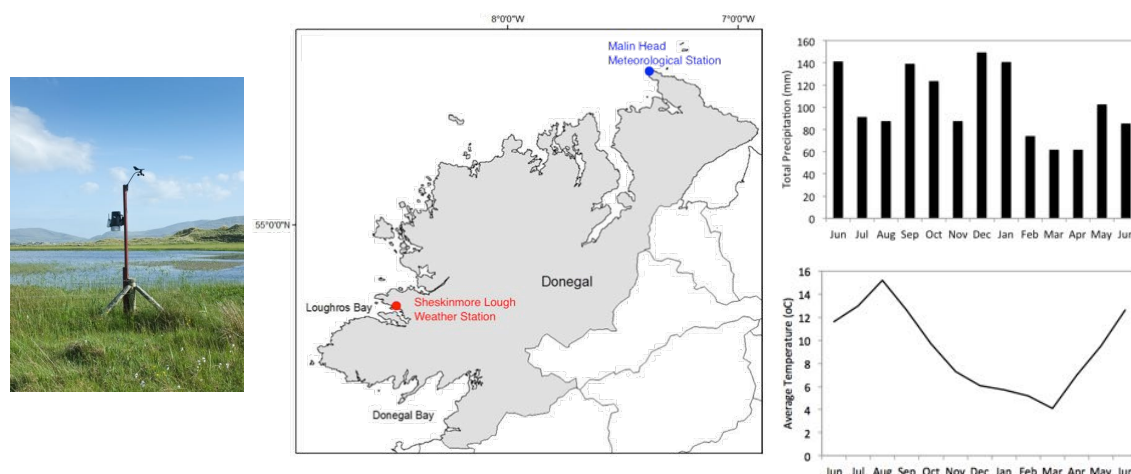


Figure 3.11 The Davis Weatherlink VantagePro2 meteorological station installed in June 2012, locations of the Malin Head meteorological station and Sheskinmore Lough weather station in County Donegal and the monthly total precipitation and monthly average temperature at Malin Head downloaded from the Met Éireann website for the June 2012 to June 2014 period.

During the latter half of 2013 and the first half of 2014, periodic communication failure between the weather station and the console resulted in a series of gaps in the observed data amounting to 53 days missing from the overall record (Table 3.2). It was unclear what the cause of the communication issues was as the signal strength had remained consistently high up to this point, but after re-positioning the console (which had no bearing on measurements taken) and replacing batteries, sufficient signal strength was regained. The data gaps were unfortunately not insignificant and the daily measures derived for the full time series were therefore filled using mean daily measures (temperature and total daily precipitation data) recorded by Met Éireann at the Malin Head weather station adjusted for the region around Sheskinmore Lough.

Table 3.2 Gaps in meteorological data observed during the second year of the monitoring period (18th June 2013 to 17th June 2014).

Date	Duration
13 th August 2013	1 day
19 th - 20 th September 2013	2 days
26 th - 27 th September 2013	2 days
4 th - 21 st January 2014	18 days
6 th - 13 th February 2014	8 days
25 th - 30 th March 2014	6 days
6 th April	1 day
19 th April - 1 st May 2014	13 days
26 th - 27 th May 2014	2 days
Total	53 days

3.3.3 Water sampling

Water samples were collected from all of the discrete period hydrological monitoring locations in June 2013. Samples were kept cool in the field using a cool box and were later

filtered using a filtration kit complete with Whatman GF/F glass microfibre filters (0.7µm pore size). They were then frozen, stored and transported as both filtered and unfiltered subsamples in a portable freezer. Once defrosted, the unfiltered water samples were analysed in the water chemistry laboratories at UCL for pH and conductivity using HACH HQ30d flexi probes. The defrosted unfiltered and filtered samples were analysed for soluble nitrate by reducing nitrate to nitrite through exposure to cadmium chips. Nitrite was then combined with reagents (Ammonium chloride, Borax, N-1-naphthylethylene diamide dihydrochloride, Hydrochloric acid, Spongy cadmium, Standard nitrate solution, Sulphanilamide) to form a red/pink azo dye, which was measured spectrophotometrically using a HACH LANGE DR5000 Spectrophotometer (EPA, 1993a). The samples were then analysed for soluble reactive phosphorus. Phosphate was combined with reagents (Ammonium molybdate, Ascorbic Acid, Potassium antimonyl tartrate, Standard phosphate solution, Sulphuric acid) to form a blue azo dye, which was then measured spectrophotometrically (EPA, 1993b). Finally, the samples were analysed for total phosphate by combining phosphate with reagents (Ammonium molybdate, Aqueous phenolphthalein indicator, Ascorbic Acid, Potassium antimonyl tartrate, Potassium persulphate, Sodium hydroxide solution, Standard phosphate solution, Sulphuric acid) to form a blue azo dye, which was also measured spectrophotometrically (EPA, 1993b).

3.3.4 Flow gauging

Discharge was measured in June 2013 in the two inflow channels and the outflow channel using salt dilution gauging as outlined by Moore (2004a). The constant rate injection method was applied in the inflows as it allows for greater accuracy in streams with low flows; velocities in these inflows were sometimes too small for reliable measurement using a conventional flow meter. Day (1977), Johnstone (1988) and Moore (2004b) claim that salt dilution gauging can be precise to within 5%. Nonetheless, Moore (2004b) claims this figure is equivalent to the accuracy of flow meter measurement; therefore flow was also measured in the inflows and outflow using a Starflow Ultrasonic Doppler flow logger (which only became available for use late in the project) and a Valeport BFM001 open channel flow meter to provide a means for comparison to validate the accuracy of the salt dilution method. Flow was measured at 0.5m intervals across the width of each channel to account for potential bankside and vegetation lag effects. Efforts were made to ensure flow was measured on several different occasions in June 2012, June 2013 and also in March 2014 to capture the full range of water levels and therefore to ensure subsequent discharge calculations were as accurate as possible.

Constant rate salt dilution involves the injection of a tracer solution (in this case salt or sodium chloride (NaCl) solution (1kg NaCl in 10L stream water)) into a stream channel at a defined rate (litres (L) per second (s)):

$$q(\text{L/s})$$

Equation 3.3

A constant rate of injection (0.83ml/s) was achieved using a Mariotte bottle constructed from a 10L carboy and funneled spigot, with a hose attached to allow for the solution to be

injected into the centre of the stream channel (Moore, 2004c). The injection rate (q) was calculated several times in the field prior to and following salt dilution to ensure consistency throughout the procedure. Using a graduated 2L cylinder and stopwatch, the time taken for the injection solution to fill the cylinder was measured three times and the average rate calculated.

Moore (2004b) states that at some distance downstream of the injection site the tracer will become uniformly mixed within the water column. Moore (2004b) suggests that this distance is approximately 25 stream widths; however this approximation is highly dependent on flow and stream morphology. A series of trial salt injections were therefore necessary to determine the actual uniform mixing distance. The conductivity of trial injections was therefore measured at regular intervals 25m, 35m, 45m and 55m downstream. As the flow velocities within the inflow channels are generally very low and their widths (2-3m), cross sections and bank morphologies relatively uniform, the conductivity of trial injections was measured towards the lower end of the Moore (2004b) distance guide of 25 stream widths. In addition to this, Moore (2004b) states that after a certain period of time, a constant rate will be achieved whereby the relative concentration (RC) of the salt tracer in the stream reaches equilibrium:

$$RC_{ss} = \frac{q}{q + Q} \approx \frac{q}{Q} \quad (q \ll Q)$$

Equation 3.4

where Q is the stream discharge (L/s) and RC_{ss} is the relative concentration at steady state (L/L).

In the case of this study, a distance of 25m was deemed optimum as a series of injection trials within the inflows indicated that low flow and vegetation patches inhibited tracer movement downstream over longer distances and tracer mixing was inadequate over shorter distances. Prior to injection the electrical conductivity (EC) and in-stream temperature were recorded at both the injection site and the downstream site using a HACH HQ30d conductivity probe. The tracer solution was then injected at a constant rate into the stream channel using the Mariotte bottle. At the same time EC was recorded at 10-second intervals 25m downstream until a relative concentration at a steady state was achieved.

As EC is linearly related to RC for dilute solutions, discharge was calculated using Equations 3.5 and 3.6; therefore RC_{ss} was determined from EC measurements:

$$Q = \frac{q}{RC_{ss}}$$

Equation 3.5

$$RC_{ss} = k(EC_{ss} - EC_{bg})$$

Equation 3.6

where k is the slope of the relation between RC and EC , and EC_{bg} and EC_{ss} are the electrical conductivities of stream water at background levels and at steady state.

Equations 3.5 and 3.6 were then combined into the following equation to calculate discharge:

$$Q = \frac{q}{k(EC_{ss} - EC_{bg})}$$

Equation 3.7

To apply Equation 3.7, however, k had to be derived. This was achieved by firstly measuring the injection rate of the salt tracer solution (q) and the background steady state values of EC and, secondly, constructing a calibration curve of RC versus EC . A secondary calibration solution was therefore created by mixing X mL of injection solution (10 mL in this case) with a measured volume (V_0) (1000 mL in this case) of stream water. This solution had a relative concentration (RC_{sec}), which was calculated as:

$$RC_{sec} = \frac{X}{V_0 + X}$$

Equation 3.8

In this case, the 10 mL injection solution (X) and a measured stream water volume (V_0) of 1000 mL produced a relative concentration (RC_{sec}) of 9.90×10^3 (L/L).

To create the calibration curve, a calibration tank was constructed using a clean watertight vessel. Stream water was measured into the calibration tank using a beaker, which was immersed in stream water at the stream edge to ensure temperatures remained constant during the calibration process. The initial conductivity (EC_0) of the stream water in the calibration tank was measured, a figure which will correspond to $RC = 0$. A known amount of secondary solution (10 mL in this case) was then added to the calibration tank using a pipette. Separate pipettes for the injection and secondary solutions were used to avoid contamination. The solution was mixed thoroughly and the EC and temperature recorded using a HACH HQ30d conductivity probe. The process of adding 10 mL of secondary solution was repeated multiple times until the EC in the calibration tank exceeded EC_{ss} . At each step the relative concentration (RC) was calculated using:

$$RC = \frac{RC_{sec} \Sigma y}{(V_c + \Sigma y)}$$

Equation 3.9

where Σy is the cumulative amount of secondary solution (mL) added to the calibration tank.

RC and EC data were then used to plot calibration curves and the best-fit lines were used to determine the slope of the curve (k) from which discharge was calculated for the two inflowing rivers using Equation 3.7 (Figure 3.12). The discharge calculated resulted in a flow volume of 5.5×10^{-5} (m^3s^{-1}) in the Duvoge river and 5.5×10^{-5} (m^3s^{-1}) in the Abberachrin river, with water levels at 0.55 m and 0.50 m above the ground surface, respectively (Table 3.3).

The outflow discharge was also measured using salt dilution gauging; however greater flow velocities and channel area meant that the slug injection method as outlined by Moore (2005) was deemed more appropriate. This method involved injecting a volume of

salt (NaCl) solution (in this case 2kg of NaCl in 10L of stream water) as a near-instantaneous slug or gulp into the centre of the outflow upstream of the sluice. At the same time, the change in conductivity was recorded at 10-second intervals using a HACH HQ30d conductivity probe at a point 25m downstream from the injection site until background levels were resumed. 25m was chosen as the optimum distance as a series of trial runs determined that the tracer had been sufficiently mixed across the channel water column at this point and in the ideal time of under 20 minutes (Moore, 2005). Prior to injection, the background conductivity and temperature of the stream were recorded at both the injection and measurement site. The conductivity of the tracer solution was also measured prior to injection and, as before, flow was also measured in the outflow using a Starflow Ultrasonic Doppler flow logger to provide a means for comparison to validate the accuracy of the salt dilution method.

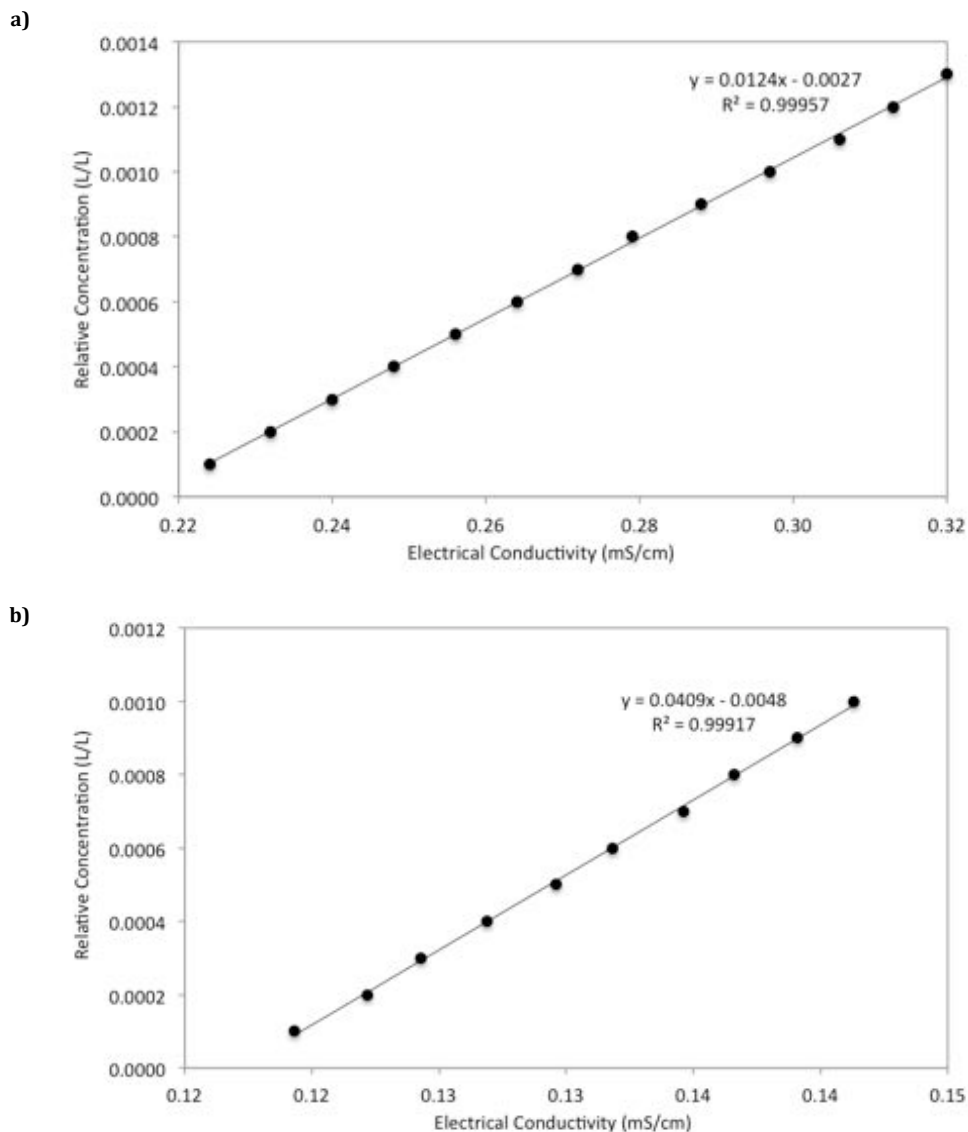


Figure 3.12 Calibration curves used to determine the slope of the curve (k) to calculate discharge in the a) Duvoige inflow (L1) and the b) Abberachrin inflow (L2).

Table 3.3 Discharge and stage in the Duvoge inflow (L1) and the Abberachrin inflow (L2) following constant rate salt injection in June 2013.

River	Discharge (m ³ s ⁻¹)	Stage (m)
Duvoge (L1)	5.5 x 10 ⁻⁵	0.55
Abberachrin (L2)	4.5 x 10 ⁻⁵	0.50

As the injection solution leaves the injection vessel, it stretches downstream as a ‘cloud’ of salty water in a process called ‘longitudinal dispersion’ (Moore, 2005). The cloud consists of a leading edge with relatively low NaCl concentrations, a central zone of high concentrations and a trailing edge of decreasing concentrations. Longitudinal dispersion therefore reduces the electrical conductivity peak as it travels downstream. As a result, the time required for the peak of the cloud to move past the downstream observation point depends inversely on the mean velocity of the stream flow. Similarly, the time it takes for the salt cloud to pass the observation point depends on the amount of longitudinal dispersion, which in turn depends on the stream velocity. A series of equations were therefore used to calculate the discharge, as at any time (t) during the salt cloud passage, the discharge of the tracer solution ($q(t)$) (Ls⁻¹) at the measurement point will be approximated by:

$$q(t) = Q \cdot RC(t) \quad \text{Equation 3.10}$$

where Q is the stream discharge (Ls⁻¹) and $RC(t)$ is the relative concentration of the tracer solution (L⁻¹) in the flow at time (t). The equation assumes that $q(t)$ is much smaller than Q .

The conductivity recorded indicated that the tracer was present throughout the duration of the salt cloud passage. In addition, the flow meter showed that stream velocity remained constant over that time therefore the following equation was applied:

$$V = \int_T q(t)dt = Q \int_T RC(t)dt \quad \text{Equation 3.11}$$

where T represents the duration of the salt cloud passage (s).

Equation 3.11 was then rearranged to solve Q :

$$Q = \frac{V}{\int_T RC(t)dt} \quad \text{Equation 3.12}$$

$RC(t)$ was then determined at the downstream measurement point at the discrete time interval Δt (in this case 10 seconds) using:

$$\int_T RC(t)dt \approx \sum_n RC(t)\Delta t \quad \text{Equation 3.13}$$

where n is the number of measurements during the passage of the salt cloud.

The relative concentration was then determined from EC :

$$RC(t) = k[EC(t) - EC_{bg}] \quad \text{Equation 3.14}$$

where $EC(t)$ is the electrical conductivity measured at time t , EC_{bg} is the background electrical conductivity of the stream and k is a calibration constant.

As k depends on the salt concentration of the injection solution and the chemical characteristics of the stream water, Equations 3.12, 3.13 and 3.14 were combined to calculate discharge:

$$Q = \frac{V}{k\Delta t \sum_n [EC(t) - EC_{bg}]} \quad \text{Equation 3.15}$$

where V is the volume of the injected salt solution.

To complete Equation 3.15, however, the calibration constant (k) had to be determined. This was achieved by plotting the relative concentration and EC values and the resultant best-fit line was used to define k for the outflow (Figure 3.13). Discharge was then calculated resulting in a flow volume of $1.1 \times 10^{-5} \text{ (m}^3\text{s}^{-1}\text{)}$ in the outflow, with water levels at 0.50m above the ground surface (Table 3.4).

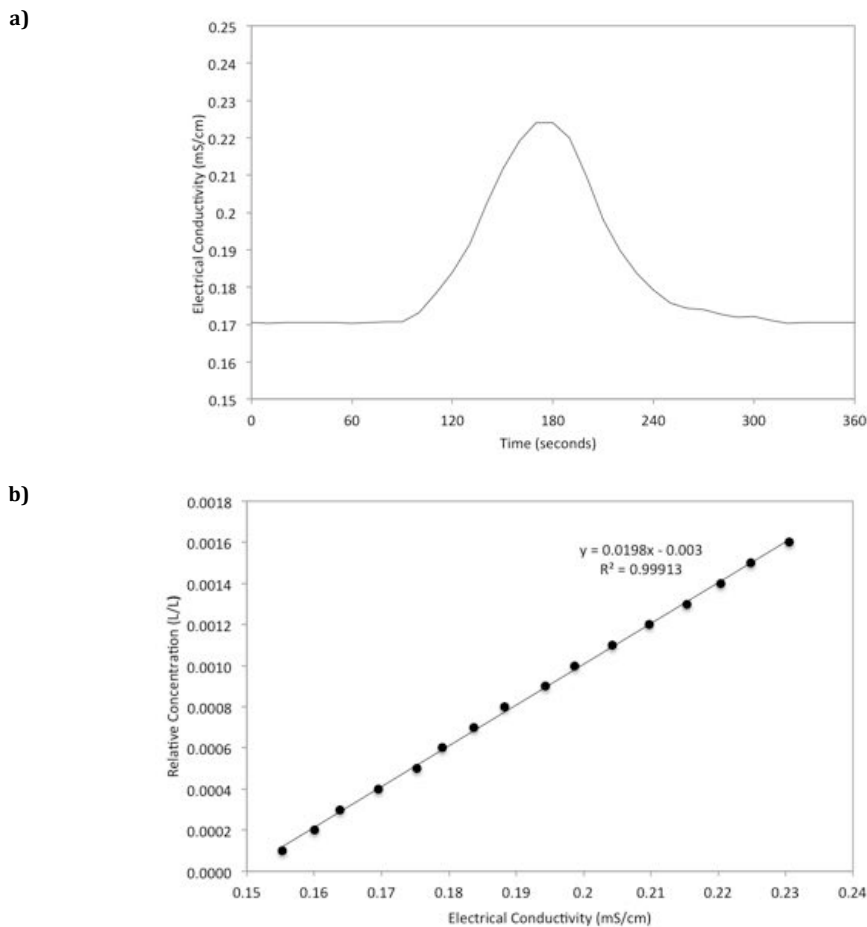


Figure 3.13 The a) downstream electrical conductivity (EC) over time following upstream slug injection of the salt solution and the b) calibration curve used to determine the slope of the curve (k) and calculate discharge in the outflow (L4).

Table 3.4 Discharge and stage in the outflow (L4) following constant rate salt injection in June 2013.

River	Discharge (m^3s^{-1})	Stage (m)
Outflow (L4)	1.1×10^{-5}	0.50

3.3.5 Topographical survey

Lake bed bathymetry, wetland surface topography and channel cross sections were surveyed in June 2013 using a differential Global Positioning System (dGPS) comprising a Leica GPS1200 base station receiver and portable Leica GS15 rover. All dipwells and TROLL loggers were levelled using the dGPS system. Topographic surveying was necessary because existing satellite-derived elevation models for the area are poor resolution (spatial and vertical) and lack bathymetric detail, and high resolution airbourne elevation surveys such as Lidar have not been undertaken in northwest Ireland to date (Schmidt & Persson, 2003; Rayburg et al, 2009; Baptista et al, 2008; Jones et al, 2012).

A permanent benchmark was set up on a secure wooden post on a high point in the grounds of the NPWS Field Centre at McGlinchys 700m to the southeast of the lake near the weather station, using nearby previously triangulated benchmarks set up for previous projects (Burningham, 1999; Barrett-Mold, 2013). The base station receiver was installed at this benchmark and used to process real-time data obtained with the rover, providing an average 3D coordinate quality of 1-2mm. The dGPS rover was attached to a backpack when surveying the wetland, and mounted on a pole when surveying the lake. A small boat was used to facilitate bathymetric measurements in the lake and elevation was measured at discrete points along transects spaced approximately 20m apart (Figure 3.14). In the wetland, the rover was set to automatically measure every metre in the horizontal axis (distance over the ground) and every 20cm change in the vertical axis (elevation). This sampling strategy was employed in parallel transects spaced approximately 20m apart. Points were also measured at spot heights and breaks of slope, as well as along morphological outlines such as around the edge of the lake.



Figure 3.14 Points collected during the topographic and bathymetric survey conducted in June 2013 (Adapted from Maher-McWilliams, 2013).

Data were additionally post-processed in Leica GeoOffice and transformed to Ordnance Survey Ireland (OSI) Irish Grid using Malin Head (i.e. mean sea level of the tide gauge at Malin Head, County Donegal) as the vertical datum (Maher-McWilliams, 2013). All GIS analysis was undertaken using the Irish National Grid (TM 65) projectio. A digital elevation model (DEM) was generated within a Geographic Information System (GIS) using ESRI ArcMap 10.1 using the methods outlined by Maher-McWilliams (2013). The surveyed DEM was also combined with the existing regional 30m-resolution Irish Grid DEM provided by OSI to ensure topographical data coverage across the wider catchment.

3.4 Paleolimnology

Multiproxy studies are frequently used in the study of paleolimnology, especially in the coastal environment (Gehrels et al, 2001; Woodward et al, 2005; Nichol et al, 2007). The use of a variety of complimentary paleolimnological techniques enables a more secure reconstruction of environmental change, and characterisation of the multiple processes and driving factors affecting the lake and wetland system (Freund et al, 2004). Similarly, a multiproxy approach was adopted to fully encapsulate changes occurring both spatially and historically. The methods used were chosen on the basis that they would most effectively capture such factors and changes, and also for compatability in terms of their ability to analyse samples taken from a sand-dominated environment. For instance, this study recognises that diatom sampling on its own is unlikely to produce a complete environmental record as diatoms are often poorly preserved within sandy sediments; however, when combined with other methods, their benefits outweigh such potential disadvantages. This is especially true when considering that the proportion of organic content contained within the majority of the sediment cores collected from the lake and wetland system was sufficient enough for adequate diatom preservation and representation. Nevertheless, diatom preservation was not consistent throughout the length of the cores. Macrofossils were therefore also identified to capture specific features of the environmental change not represented within the diatom record.

Sediment core samples were collected from the lake and wetland during June 2012 and June 2013. The aims of collecting short sediment core samples in June 2012 were to: explore sediment stratigraphy patterns across the lake and wetland system; determine the preservation potential of paleological indicators; and develop an initial understanding of the recent past ecohydrology of the system. The purpose of extracting two full-length sediment cores from the lake in June 2013, once it became clear that paleoecological indicators were adequately preserved, was to provide a more complete understanding of the past ecohydrological development of the system.

3.4.1 Exploratory cores

3.4.1.1 Geochemistry

During dipwell construction in June 2012, exploratory sediment cores were extracted during an exploratory study in discrete sections of approximately 25cm using a soil auger. At some of the hydrological monitoring sites, samples were collected from the sediment layers evident within the augured sections brought to the surface (Figure 3.15). The depths of the layers were measured and recorded, from which sediment stratigraphies were created for each sampled dipwell. Description of the sediment layers included colour, organic content (including the type of plant remains where possible) and texture. Exploratory sediment cores were also extracted from the lake surface by inserting a PVC dipwell tube into the surface sediments. The majority of the exploratory cores (all excluding W1 and W4, as sediment samples were not retained due to contamination) were analysed for both their organic and carbonate content using the Loss-On-Ignition (LOI) method. LOI, which is based on the percentage weight lost when dried samples are placed in a furnace set at 550°C for at least 2 hours, was used to produce crude measurements of organic content (Dean, 1974; Heiri et al, 2001). Similarly, the sediment carbonate content (i.e. the amount of carbon dioxide lost through conversion to oxides) was determined using the same method, only this time the post-LOI samples were heated to 925°C and then re-weighed (Dean, 1974; Heiri et al, 2001).

3.4.1.2 Particle size analysis

Grains of silica, found within all of the exploratory sediment samples collected from the lake and dipwells, were analysed to determine particle size in order to consider variations in sediment source, environment of deposition and mode of sediment transport. Sub-samples were weighed and heated in a furnace to 550°C to remove any organic matter. The samples were then reweighed and the organic content recorded to provide a comparison for the previous LOI analyses (Chester & Hughes, 1967). The samples were washed with 10% hydrochloric acid and left overnight to ensure the complete dissolution of any calcium carbonate present within the sediment. The samples were then repeatedly centrifuged and rinsed with distilled water until all acid residues were removed and particles disaggregated. The samples were placed in a drying cabinet overnight and, once dry, were reweighed to determine their calcium carbonate content. Finally, the particle size of each sample was analysed by optical laser diffraction, specifically using a Malvern MasterSizer 2000 particle analyser with a Hydr2000 MU pump and cell; this system can address the size range of 0.02 to 2000µm to be measured. Laser diffraction, a commonly used method of obtaining particle size distributions, measures the refractive indices of light scattered by a sample in order to calculate particle size. Results are dependent on the refractive indices of the material and medium into which it is suspended; therefore the particle size distribution obtained depends critically upon assumptions made about the optical properties of the study materials (Wedd, 2003). The conversion from the light scattering data to particle size distributions uses the Mie theory, which assumes all

particles are spherical (Campbell, 2003). The minerogenic sediment within the Loughros More and Sheskinmore Lough system is dominated by quartz grains which are largely equidimensional and relatively consistent in colour, thereby presenting few complications in laser diffraction grain size analyses.



Figure 3.15 Locations of the exploratory sediment cores collected during the exploratory study in June 2012 from hydrological monitoring sites across the wetland system (S3, W1, W2, W3, W4, W6, W7) and from within Sheskinmore Lough (L3, L3.2, L3.6) and Sandfield Lough (L6).

A standard operating procedure (SOP) was set up and used for all samples so the conditions of each measurement would be the same. A silica standard (24/013W) of known grain size composition was measured to ensure accuracy; there was found to be no significant difference between the known and measured values (p -value = 0.007). Initial examination of the sedimentology of the samples showed the majority of grains to be quartz sand grains, therefore within the SOP the particle refractive index of and particle absorption index for silica, 1.544 and 0.01 respectively, were entered for the conversion to grain size data calculations. The dispersant used was tap water and therefore a dispersant refractive index of 1.33 was entered into the SOP. An appropriate amount of sample was placed into 500ml of tap water until a suitable laser obscuration was reached (i.e. between 10% and 20%); this amount varied depending on the sample but was approximately 2g. The pump speed was set to 2000rpm and the ultrasonic stirrer used for 10 seconds to disaggregate the sample before measurements were taken. Three measurements of a count time of 30 seconds per sample were made with a 10 second gap between measurements. The output data is in the form of volume of sample between size classes. The particle grain size fractions of the samples were analysed using GRADISTAT to

examine the grain size distributions, statistics and textural descriptions (Blott & Pye, 2001). The working unit for sediment size analysis in this study is phi (Φ), as converting the geometric size scale to an arithmetic size scale via log transformation aids the statistical analysis of the sediment size distribution data (Hobson, 1979).

3.4.1.3 Macrofossils

The exploratory sediment samples collected from the lake and dipwells in June 2012 were prepared for macrofossil analysis. Approximately 50cm³ of each sample was washed through 125 μ m and 355 μ m sieves. The 355 μ m sample was examined in its entirety, but only a quarter of the 125 μ m sample was examined, as macrofossil specimens were sparse in this division. Small quantities of each sample residue were dispersed in 2–3 mm depth of water in a Petri dish and all plant remains were picked out systematically at 10–40x magnification under a binocular dissecting microscope. Macrofossils were identified using a reference collection of plant parts, as well as seed atlases (Berggren, 1964; Berggren, 1981), photographs and illustrations in publications including Jessen (1955), Grosse-Brauckmann (1986), Preston (1995), Canellas-Bolta et al (2012) and Qiang et al (2013). All macrofossils are presented as numbers per 100cm³ of sediment. Although macroremains in surface sediments are not strictly macrofossils they are referred to as such here to simplify terminology.

Analyses of floral and faunal macrofossils (seeds, spines, animal remains) have been conducted in Quaternary studies for decades to provide information on both aquatic and terrestrial plant and animal taxa proximal to lakes and wetlands (Birks, 1973; Birks, 1980). In particular, research has shown that macrofossil analysis can be used to directly reconstruct changes in the macrophyte community composition within lakes (Brodersen et al, 2001; Rasmussen & Anderson, 2005; Reid et al, 2007). Macrofossils are more likely to provide a consistent amount of taxonomic resolution over time compared to historically recorded data (Davidson & Jeppesen, 2013). In addition, comparisons of the lake sediment record with historically recorded data within previous studies have demonstrated that the analysis of macrofossil remains can provide reliable information on past environmental change (Davidson et al, 2005; Salgado et al, 2010). Although rare species are often not reliably recorded within the sediment record, their limited presence may still be sufficient enough to describe broad historical fluctuations (Heino & Soininen, 2010). Thus, it may be possible to determine the degree of change from rare species or reference taxa at a single lake site (Davidson & Jeppesen, 2013). It should also be mentioned that the reconstruction of past lake-level changes, and hence past climates, can best be achieved by examining macrofossils (Gaillard & Digerfeldt, 1991; Harrison & Digerfeldt, 1993; Hannon & Gaillard, 1997). Integration of macrofossil evidence with other paleolimnological proxies and with modern spatial and timeseries data can therefore help build extremely detailed pictures of past environmental change.

3.4.1.4 *Diatoms*

The exploratory sediment samples collected from the lake and dipwells in June 2012 were also prepared for diatom analysis. Hydrogen peroxide (H_2O_2) was applied to the samples suspended in a water bath to remove organic matter or any mineral material present that could ultimately reduce or inhibit diatom identification (Renberg, 1990). In this study 5mls of 30% H_2O_2 was added to a test tube containing approximately 0.1g of wet sediment. This was then placed in a water bath at room temperature. If the sediment did not react violently the temperature was increased to 80°C and left for 2 to 4 hours until all organic material had been removed. The tubes were then removed from the bath and 1 to 2 drops of 50% Hydrochloric acid (HCl) added to eliminate any remaining H_2O_2 and carbonates. Samples were then centrifuged at 1200rpm for 4 minutes and the supernatant decanted off. They were then topped up with distilled water and the washing process repeated at least 3 times to ensure the complete removal of residue H_2O_2 and HCl; 1 or 2 drops of weak ammonia (NH_3) solution were added to each sample with the final wash to ensure clays remained in suspension and to prevent diatoms clumping together during slide preparation. Samples were then diluted to a concentration of 0.1g of diatom sample in 10ml of water. Once in suspension, 1ml pipettes were used to place 0.5ml of each well-mixed diatom sample onto each cover slip, which were then left to dry overnight, thereby ensuring an even spread of diatoms over the cover slip. Once dry, glass slides were laid out on a hotplate set to 130°C and each sample cover slip was placed onto the slide and sealed with a drop of Naphrax. Slides were then left to cool for 15 minutes before being stored in a protective slide case.

Slides were viewed under the microscope to assess whether the sample concentration was suitable for identification. As the initial concentration of 0.1g of diatom sample in 10ml of water yielded a diatom assemblage far too sparse for identification, the concentration was increased to 1g of diatom sample in 10ml of water and a new set of slides prepared for identification microscopy. Diatom species were identified and their abundance recorded by scanning across the cover slip in a series of horizontal and vertical transects. The valves were counted under phase contrast illumination at x1000 magnification. Diatom species were identified using various diatom atlases including Foged (1977), Barber & Haworth (1981) and Krammer & Lange-Bertalot (1986; 1988; 1991a; 1991b). Slides that contained a large abundance of diatoms were scanned until 500-600 individual diatoms had been counted. Slides containing moderate and low diatom abundances were scanned until 200-300 and 50-100 individual diatoms, respectively, had been counted. To obtain data on the relative abundance diatom frequencies are expressed as percentages of total diatom valves counted (TDV).

The abundance and diversity of diatom species in coastal and freshwater wetland environments makes them a valuable indicator of past and present environmental conditions. Consequently, diatom analysis is a well-established palaeoecological method used in resolving the past ecologies and hydrologies of marine, coastal and freshwater deposits (Sherrod et al, 1989; Foster et al, 1991; Sawai et al, 2008; Holmes et al, 2010; Stendera et al, 2012). In addition, diatoms are frequently used to determine Holocene

coastal geomorphological evolution (Shaw & Carter, 1994; Gehrels et al, 2001; Long, 2001). Diatoms are found wherever there is moisture and sufficient light for photosynthesis. Habitats vary significantly depending on the chemical and physical properties of the environment and, as a result, can be identified by the characteristic diatom flora they support (Hendey, 1964).

In littoral zones, diatom flora are largely determined by the properties of the substratum. For example, bedrock supports only those species that can physically attach to the surface, whereas sand substrates can also support species that form a film across the surface (Hendey, 1964). Other factors influencing the distribution of diatom species include wave action, nutrient supply, temperature, pH, salinity, vegetation, exposure to sunlight and competition with other organisms (Battarbee et al, 2001; Roe et al, 2009). Diatoms are regarded as very sensitive indicators of salinity and are often used to record palaeoenvironmental changes in salinity (Healey, 1997; Freund et al, 2004; Holmes et al, 2010).

An advantage of using diatoms as a proxy for environmental conditions is their presence throughout the year (Hendey, 1964). This increases their efficacy as an environmental proxy as it negates the problems associated with species migration, as seen in other indicator species (e.g. Foraminifera) (Roe et al, 2009). The siliceous cell walls of diatoms generally fossilize well and, as a result, are valuable palaeoenvironment indicators (Vos & de Wolf, 1993). There are issues, however, with the accuracy with which diatom assemblages preserved within sediments represent the composition of the source communities from which they were derived (Battarbee et al, 2001). For example, in some extreme cases diatom species are completely absent from lake sediment records due to dissolution problems, as preservation is largely determined by pH. In the coastal environment, one of the main problems is distinguishing between autochthonous species and transported, allochthonous species (Freund et al, 2004). This is particularly important in estuarine environments where zonation based upon this distinction is of major importance. The percentage of broken diatom frustles has therefore been used in previous studies as an indication of transportation processes and the overall energy of the environment (Freund et al, 2004; Colombaroli et al, 2007). It should also be noted that the presence of fragile species within the sediment record is a more reliable indicator of palaeoenvironment than the more heavily silicified species, as they are more likely to remain in situ after deposition.

3.4.2 Lake cores

Two sediment cores were collected from Sheskinmore Lough in June 2013 using a percussion corer (Figure 3.16). This was deemed an appropriate number as other studies have shown that past lake vegetation can be suitably represented by two cores (Davidson et al, 2010; Sayer et al, 2010). In addition, submerged vegetation within the lake exhibits low spatial heterogeneity, therefore two cores would adequately reconstruct past changes at the scale of whole lake (Sayer et al, 2010). Coring locations were chosen on the basis that one deep and one shallow core would capture both lake-wide changes in macrofossil

composition and diatom assemblages, as the latter especially, are spatially less variable in small lakes (Anderson, 1986; Sayer et al, 2010). Steel tubes (40mm diameter) were hammered into the lakebed at the two locations shown in Figure 3.16. Considerable force was required to insert the steel tubes due to the dominance of hard sandy sediment layers identified during exploratory coring. Core 1 was collected from the deepest part of the lake (>120cm depth) near the Duvoge inflow and Core 2 was collected in a shallow area (>30cm depth) on the south side of the lake. Once extracted, the cores were capped with rubber bungs, sealed and stored in a cold store until further analysis.

Core sub-sampling was conducted in the UCL laboratory in January 2014 and the stratigraphies of the two cores were also examined in situ. Cores 1 and 2, which measured 65cm and 40cm in length, respectively, were split lengthways to preserve the stratigraphic context, and individual units and layers were identified and described based on visual classification. The cores were then divided into 76 and 58 individual samples, respectively, based on this visual analysis. The measured sections encompassed either 0.5cm or 1cm of core length depending on down core stratigraphic transitions. Before any analysis was undertaken on the downcore data the effects of compression of the sequence during core collection was taken into account.

Sediment samples were transferred immediately following division into securely sealed plastic sample bags and were stored in dark refrigerated conditions to prevent further biological activity and the introduction of modern carbon contamination. Before any analysis was undertaken on the downcore data the effects of compression of the sequence during core collection was taken into account. The degree of compression was calculated by dividing the distance between the top of the core tube and the sediment by the compressed length of the core. The result was then added cumulatively to each sample point. Although compression is unlikely to have been uniform throughout the core this was seen as the most effective way of “uncompressing” the core. The two cores were analysed for both their organic and carbonate content, particle size, macrofossils and diatom assemblages using exactly the same methods as those applied to the exploratory sediment cores outlined previously.

Radiocarbon analysis (^{14}C) was conducted by the Natural Environment Research Council (NERC) AMS (accelerator mass spectrometry) Radiocarbon Facility on two samples from Core 1 extracted from the deepest parts (<1.5 m) of Sheskinmore Lough (Table 3.5). Other dating methods were considered such as ^{210}Pb and ^{137}Cs , however these methods are often problematic in very sandy environments. Radiocarbon dating was chosen as the most appropriate dating method as Shaw and Carter (1994) successfully achieved radiocarbon dates for sandy peat layers at a site on the Duvoge floodplain, adjacent to Sheskinmore Lough. Alternative dating methods such as Cs^{137} and ^{210}Pb were investigated; however poor dating precision on saltmarsh cores obtained elsewhere within Sheskinmore Lough SPA indicate that those techniques are likely to be extremely unreliable due to the abundance of sand (Barrett-Mold, 2012).



Figure 3.16 Location of the two full-length sediment cores collected from the channel feature within the lake in June 2013.

Two radiocarbon-dated samples were chosen as an optimum number as it takes into account the modest length of Core 1, the limited number of peat layers present and their down-core distribution (Figure 3.17). The strategy of dating two ‘rangerfinder’ samples was chosen as it: 1) provides a temporal context for the shifts between sand-dominated and peat-dominated activity within the lake and wetland system; 2) provides the time period since the establishment of *N. flexilis* within the lake; 3) ensures that the analysis of palaeoecological evidence is bounded by an appropriate chronology; and 4) provides a more robust context for future simulation of climate change impacts on the ecohydrology of the site. After consultation with Steve Moreton of the NERC Radiocarbon Facility (NRCF) at East Kilbride (email comm. 28th February 2014), sandy peat layers were chosen as the optimum material for dating as they contain enough organic matter for analysis and closely mark the transitions from sand-dominated to peat-dominated episodes. In addition, 1cm³ was chosen as the optimum bulk sample size to ensure that a sufficient amount of carbon would be available for analysis. Macrofossils and other identifiable terrestrial plant material, the product of a single year’s growth and likely to have a short transport history, were also provided. The precision of this analysis was a minimum error term of ± 35 years based on that reported by the NRCF for routine analysis (Steve Moreton email comm. 18th March 2014).

Table 3.5 Specification of samples submitted to the NERC Radiocarbon Facility for initial radiocarbon dating (A17 and A67) and the subsequent pair of samples (A25 and A63) submitted to the NERC Radiocarbon Facility for radiocarbon dating.

Sample code	Depth range (cm)	Sample size (cm ³)	Material	Minimum precision (+/-yrs)	Location within core
A17	12-13	1	Sandy Peat	35	Interface between sand- and peat-dominated sections
A67	59-60	1	Sandy Peat	35	Interface between sand- and peat-dominated sections
A25	17-18	1	Peat	35	Top of upper peat-dominated section
A63	55-56	1	Peat	35	Base of lower peat-dominated section

This study acknowledges that radiocarbon analysis of samples aged <1000 years has its limitations, especially in terms of calibration between the period of about 400 cal. years BP and 1950AD. It should be stressed, however, that since its formation the lake has been completely isolated from significant marine influence including incursions during storm events; therefore there should be no effect on the apparent radiocarbon age of organisms growing within the lake. Consultation with Steve Moreton (email comm. 18th March 2014) revealed that, for routine analyses, the NRCF reports a minimum error term of ± 35 years; however that error term can be reduced for recent samples if they are measured using high precision techniques. Such analyses can narrow the probable age range when conducting calibrations to convert radiocarbon years to calendar years. Although they reduce the likelihood of multiple interpretations, this study recognises that greater precision does not completely eradicate dating uncertainties. High precision analysis at the NRCF is resource-intensive and comes with increased expense, thus greater justification is required. In the context of this research, approximate dates (i.e. proposed rangefinder samples) are a first step to establishing the likely need and value of greater precision.

Consultation with Steve Moreton led this study to the conclusion that, although analysis of more samples at higher precision levels means less ambiguity within the results, lack of knowledge regarding the overall age of the cores prior to the radiocarbon analysis described above, confirms that the analysis of two rangefinder samples was the most appropriate starting point. Radiocarbon analysis of the two core samples has therefore provided direction for potential follow-up analysis after this study. The method, conducted by the NERC radiocarbon facility at East Kilbride, is as follows:

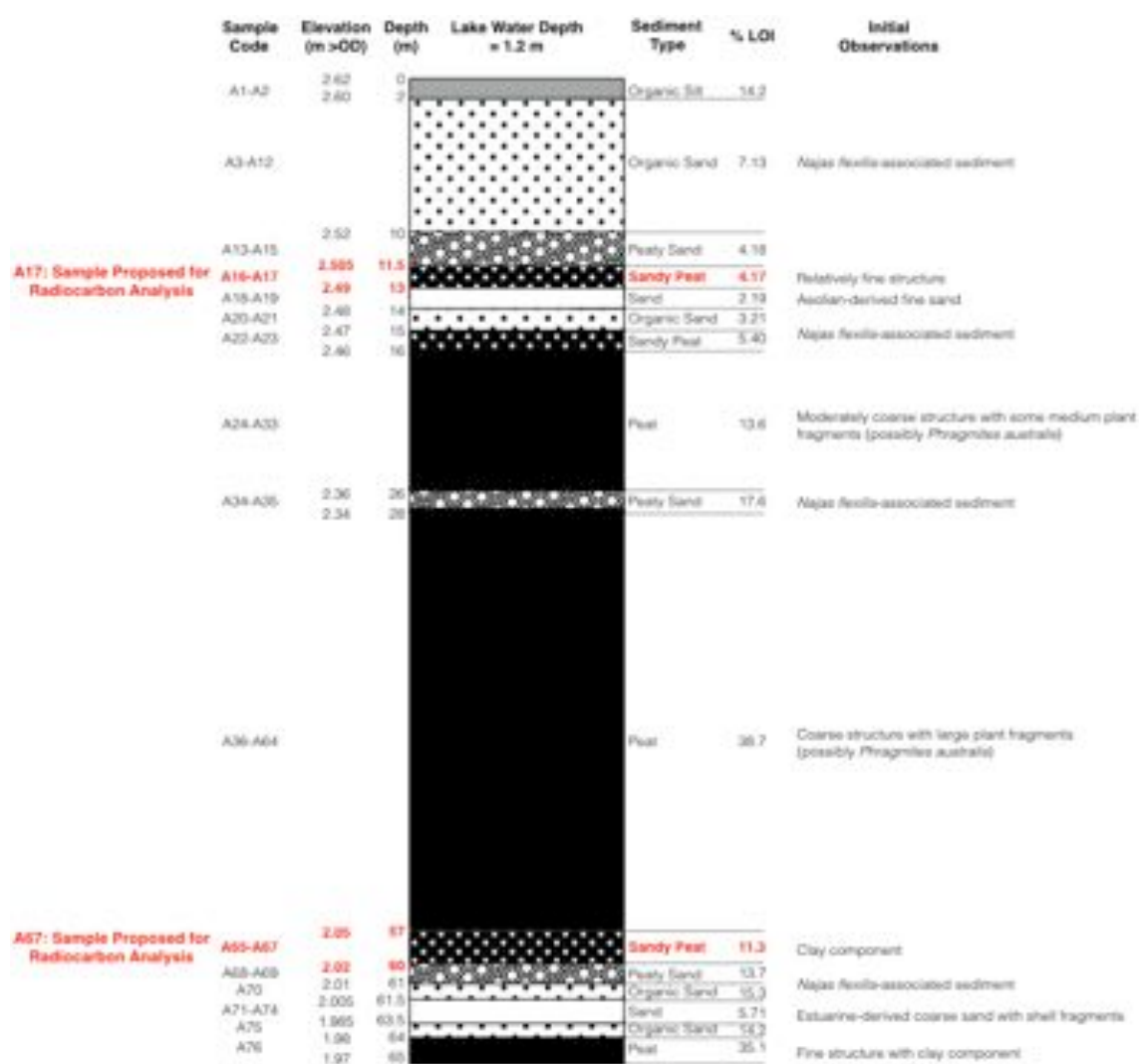
The two raw samples had to be pre-treated prior to radiocarbon analysis. Firstly, the samples were digested in 2M HCl (hydrochloric acid) at 80°C for 8 hours, they were then washed free from mineral acid with deionised water and digested in 1M KOH (potassium hydroxide) at 80°C for 2 hours. The digestion was repeated using deionised water until no further humics were extracted. The residue was rinsed free of alkali, digested in 2M HCl at 80°C for 5 hours, then rinsed free of acid, dried and homogenised. The total carbon in a known weight for each of the pre-treated samples was recovered as CO₂ by combustion

with CuO (copper oxide) in a sealed quartz tube. The gas was then converted to graphite by Fe/Zn (iron/zinc) reduction (Luz Maria Cisneros-Dozal email comm. 12th January 2015).

After being prepared to graphite during pre-treatment, the samples were passed to the SUERC AMS Laboratory for ¹⁴C analysis. In keeping with international practice, the results were reported as conventional radiocarbon years BP (relative to AD 1950) and % modern ¹⁴C, both expressed at the $\pm 1\sigma$ level for overall analytical confidence. The results were corrected to $\delta^{13}\text{C}_{\text{VPDB}}\text{‰}$ -25 using the $\delta^{13}\text{C}$ values provided in the report. The $\delta^{13}\text{C}$ values were measured on a dual inlet stable isotope mass spectrometer (Thermo Fisher Delta V) and are representative of $\delta^{13}\text{C}$ in the original, pre-treated sample material. The quoted precision is the uncertainty of repeated measurements of the same CO₂ aliquot and is therefore attributable to machine error only.

Results from the original sample submission (allocation 1791.0414) presented a problematic chronology for Sheskinmore Lough. Although comprising sandy peat, the samples contained far less organic material than optimal for AMS analysis. Furthermore, the upper layer contained coal fragments (likely wind-blown from local home fires) that generated a ¹⁴C age far older than expected. The age for the lower sample was more in line with that expected, but due to the low organic levels, it was difficult to place confidence in the dates and they offered little in terms of framing the chronology of lake-wetland development and evolution. Two further samples from the two peat-dominated units in the core were therefore submitted for dating (Table 3.5).

Position:	54.80907205N, 8.46424649W	Date extracted:	12 th June 2013
Lake depth:	1.2m	Date divided:	16 th January 2014
Core length:	65cm	Sample codes:	A1-A76



Sample A17 Depth 12-13cm



Sample A67 Depth 59-60cm



Figure 3.17 Sheskinmore Lough Core 1 stratigraphy and selected radiocarbon samples (highlighted in red), their respective depths: A17 (12-13cm) & A67 (59-60cm) and their appearance within the core (see red marks in photographs).

4 Contemporary ecohydrology of Sheskinmore Lough

4.1 Introduction

The aim of this chapter is to assess the contemporary ecohydrological conditions in conjunction with the edaphic environment of the lake and wetland system at Sheskinmore Lough. The chapter describes the abiotic and hydrological controls on species distribution and assemblages. The results contribute to wider scientific knowledge of coastal lakes and wetland systems, as research on their ecology and environmental functioning is limited (Van Groenendael et al, 1993), and in Ireland is negligible (Caffrey et al, 1999). Specifically, this chapter provides the contemporary perspective to support historical reconstructions presented in Chapter 5 and modelled futures evaluated in Chapter 6. The results presented here are derived from field campaigns in June 2012 and June 2013, and monitoring instigated in June 2012.

4.2 Ecology

Habitats within the Sheskinmore Lough SPA boundary range from coastal to terrestrial systems impacted by both aeolian and hydrological processes (Figure 4.1). The total area of the SPA is 5.63km² and sand dunes dominate, covering nearly half of that area (2.45km²; 43%). In contrast, the open water areas, including Sheskinmore Lough itself, encompass only a small proportion of the SPA (0.10km²; 2%). The broader wetland area covers 0.21km² (4%) with a further 0.27km² covered by *Phragmites australis* reedbed (5%). These three aquatic habitats (wetland, reedbed and the open water of Sheskinmore Lough), which together account for 12% (0.67km²) of the SPA, provide clear delineation between the ecological areas or 'zones' that are examined independently in this chapter. In addition to these areas, both the lake and the reedbed habitats are connected to the floodplains of the Abberachrin and Duvoge rivers that extend into valleys to the east and northeast covering a further 0.09km² (2%). Dune and backshore ponds account for a further 0.05km² (1%) of aquatic habitat. Fringing the wetland is a 0.35km² (6%) band of wet grassland that extends to Sandfield Lough in the southeast corner of the SPA. Surrounding the wet grassland habitat is machair grassland (0.16km²; 3%). A large area of peatland (1.22km²; 22%) covers the interfluvium between the Abberachrin and Duvoge valleys, and between the Sheskinmore and Sandfield Loughs. The hinterland to the immediate north of the Sheskinmore Lough basin covers 0.37km² (6%) and comprises steeply rising ground formed of a mosaic of bare rock, heath and peatland. The remaining SPA area at the western and southwestern fringe comprises Atlantic salt meadows (0.22km²), heath (0.07km²) and a dissipative beach (0.12km²), which cover 4%, 1% and 2%, respectively.

4.2.1 Wetland and reedbed macrophytes

The wetland and reedbed habitats together comprise a mix of aquatic, marginal and grassland plant species: a total of 71 species were recorded in the June 2012 and 2013 wetland surveys and the July 2014 reedbed survey (Figure 4.2). Macrophytes in the reedbed adjacent to the wetland were surveyed as a separate zone to the open water environment of the lake and the terrestrial wetland habitat to the west. The most frequently occurring species in the wetland are *Sphagnum* sp. (present in 69 of the 90 sample sites) and *Equisetum palustre* (60/155 samples). The reedbed habitat is largely reed-dominated and transitions from the large waterlogged region surrounding the northwestern side of the lake (occupied almost exclusively by *Phragmites australis*) to the western fringe of the reedbed where *Phragmites australis* phases out and is replaced by more diverse terrestrial macrophyte assemblages. *Phragmites australis* is the most abundant species, covering 25.6% of the sample area, with *Sphagnum* sp. comprising 12.9%. The second most frequent species in the reedbed, after *Phragmites australis*, is *Sphagnum* sp., which is present in 21 of the 33 sample sites. *Sphagnum* sp. is also the most abundant species in the wetland, covering 17.6% of the whole sample area. The least frequent species that also exhibit low abundance in the wetland include *Cardamine flexuosa*, *Carex paniculata*, *Listera ovata*, *Pimpinella major*, *Triglochin palustris* and *Veronica chamaedrys*, each found in only 1 quadrat. In contrast, the most infrequent species in the reedbed include *Anthoxanthum odoratum*, *Bromus commutatus*, *Carex aquatilis*, *C. bigelowii*, *C. hostiana*, *Eleocharis palustris*, *Juncus effuses*, *Poa flexuosa*, *Trifolium repens* and *Typha latifolia*, which are also each found in only 1 quadrat.

Wetland macrophyte species richness is greatest (13-24 species) in the wetland southeast of the lake and also high to the west of the wetland (13-18 species) (Figure 4.3). Low species richness (4-6 species) is found in small pockets across the west arm of the wetland in close proximity to areas of high species richness leading to a disparate pattern in this measure of plant species diversity. The greatest macrophyte species richness (16-24 species) within the reedbed is located to the north, with pockets of high species richness fringing the western edge. In contrast, the south edge of the reedbed has the lowest species richness (only 1-6 species) where *Phragmites australis*, located on waterlogged ground, is tallest (>2m) and densest (>80% cover). Species richness peaks on each transect at the boundary from reedbed to wetland. Wetland quadrats located towards the central and eastern end of the wetland nearer to the reedbed are at the lowest elevations (between 3.01mOD and 4.00mOD). The species here represent a mixture of plants characteristic of both dry (including: *Anthoxanthum odoratum*; *Bromus commutatus*; *Cynosurus cristatus*; *Holcus lanatus*; *Leucanthemum vulgare*; *Poa flexuosa*; *Trifolium campestre*; and *Salix repens*) and wet habitats (including: *Carex acutiformis*; *Dactylorhiza incarnata*; *Equisetum hyemale*; *Menyanthes trifoliata*; *Pinguicula vulgaris*; *Phragmites australis*; *Ranunculus sceleratus*; and *Schoenus nigricans*). Intermediate elevations (between 4.01mOD and 5.50mOD) are characterised by the highest incidences of species richness ($R^2 = 0.79$; $p = <2.2e^{-6}$) with two distinct groups of species comprising species of grassland habitats (including: *Carex flacca*, *Molinea caerulea*, *Lolium perenne*, *Potentilla*

tabernaemontani and *Veronica chamaedrys*) and those wet habitats (including: *Agrostis capillaris*, *Equisetum fluviatile*, *Iris pseudacorus*, *Sparganium erectum* and *Typha latifolia*). The spatial pattern of species richness in the reedbed is reflected in the percent cover of *Phragmites australis* with the majority of the highest percentages (51-100%) found towards the east and south within the main body of the reedbed where species richness is lowest ($R^2 = -0.74$; $p = <2.7e^{-10}$). The lowest *Phragmites australis* cover (5-30%) is found around the northern and western fringe of the reedbed where species richness is highest.

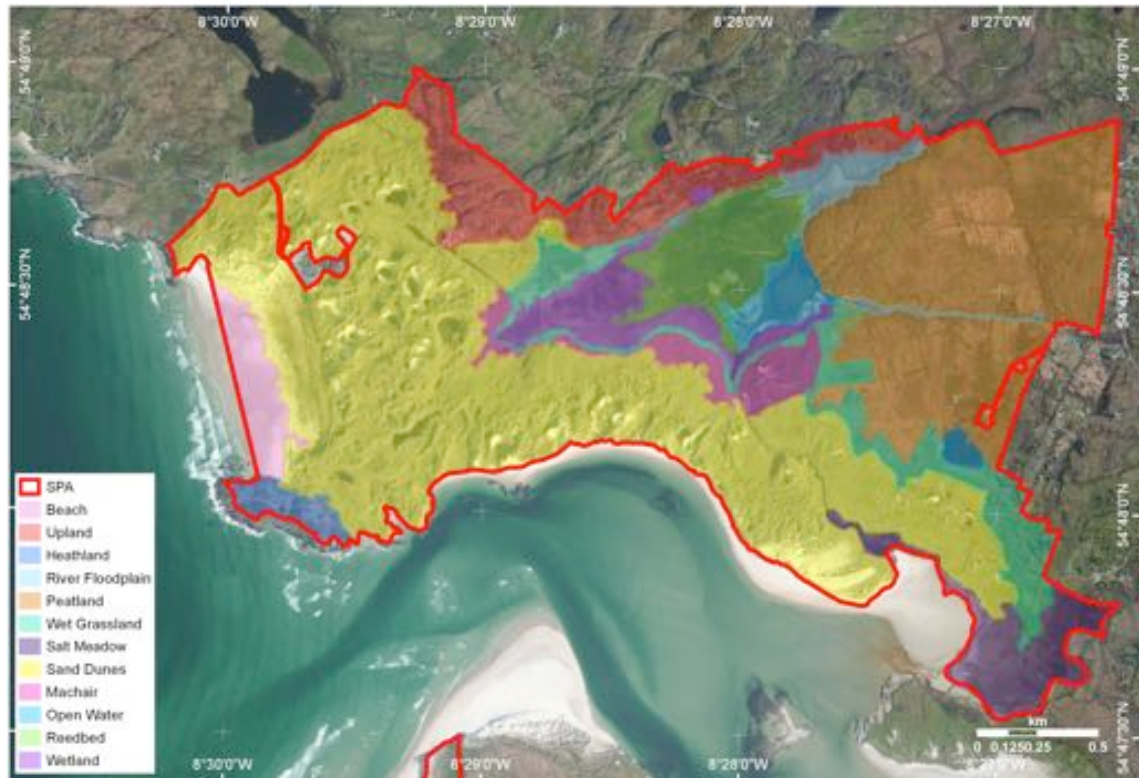


Figure 4.1 Spatial distribution of habitat areas within Sheskinmore Lough SPA in 2012. The habitats were digitised and their areas calculated in ArcMap 10.1 using a combination of Bing (November 2011 to March 2012) satellite imagery, on-the-ground observations and GPS data (Bing, 2013).

Transect locations were systematically chosen in the field to extend across the *Phragmites australis* stand as it existed in July 2014. The habitat areas in Figure 4.3, however, were digitised using a combination of satellite imagery, GPS data and on-the-ground observations from winter 2011 to summer 2012. The 2014 quadrat positions illustrate the expansion over two years of the reedbed, specifically *Phragmites australis*, into what was formally wetland habitat in 2012. Change in the reedbed to wetland boundary is delineated in Figure 4.4, showing expansion is greatest in an overall westerly direction with the two leading ‘arms’ embracing the south and north sides of the wetland. The reedbed has also expanded to the north and northwest and is beginning to fill the area at the foot of the upland habitat. Also evident in the satellite imagery are the pockets of low *Phragmites australis* cover in waterlogged hollows between mounds of higher and drier ground.

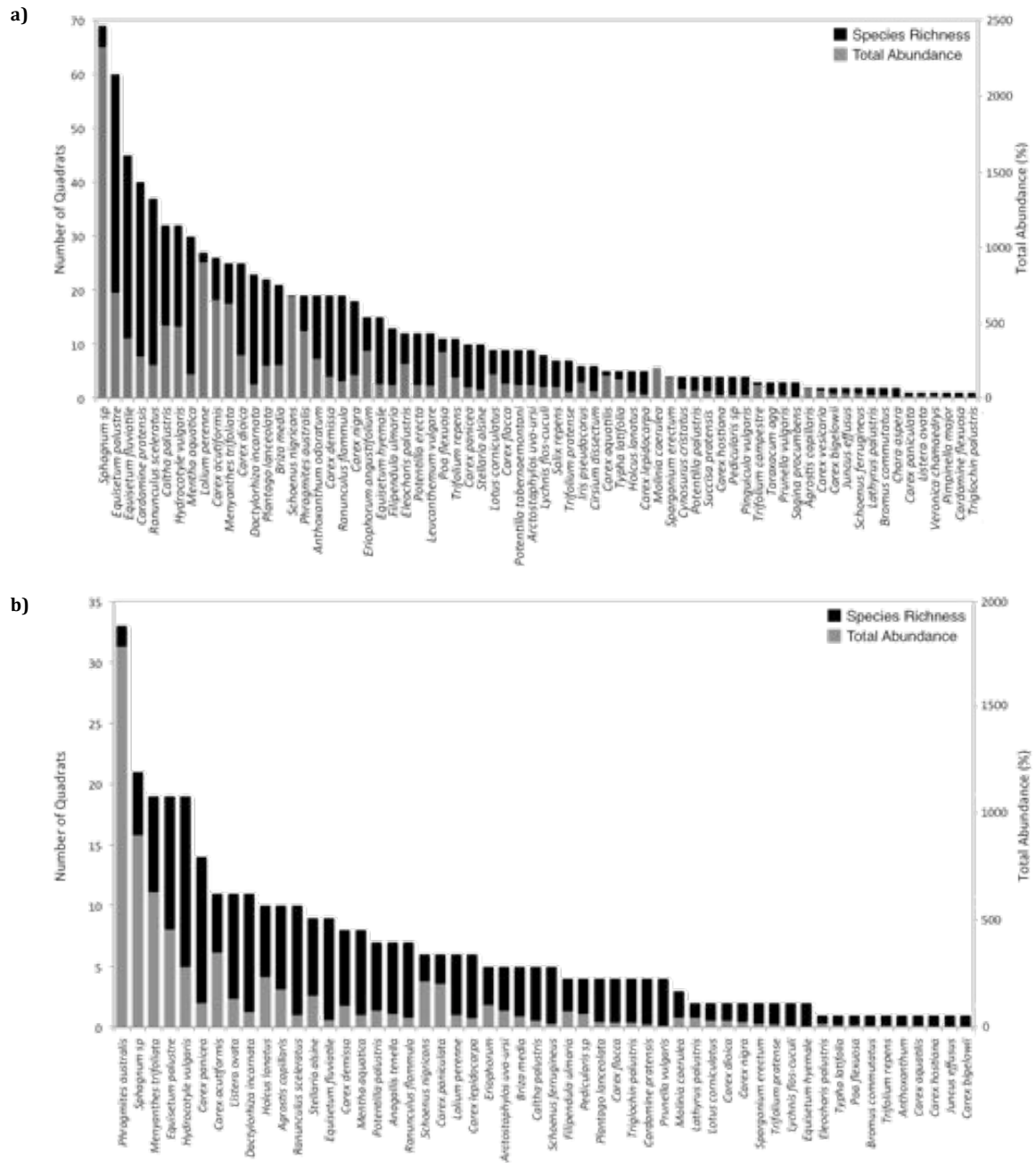


Figure 4.2 Emergent macrophyte species richness and total abundance identified a) within the 90 quadrats surveyed in the wetland in June 2012 and 2013, and b) within the 33 quadrats surveyed in July 2014 in the reedbed habitat.

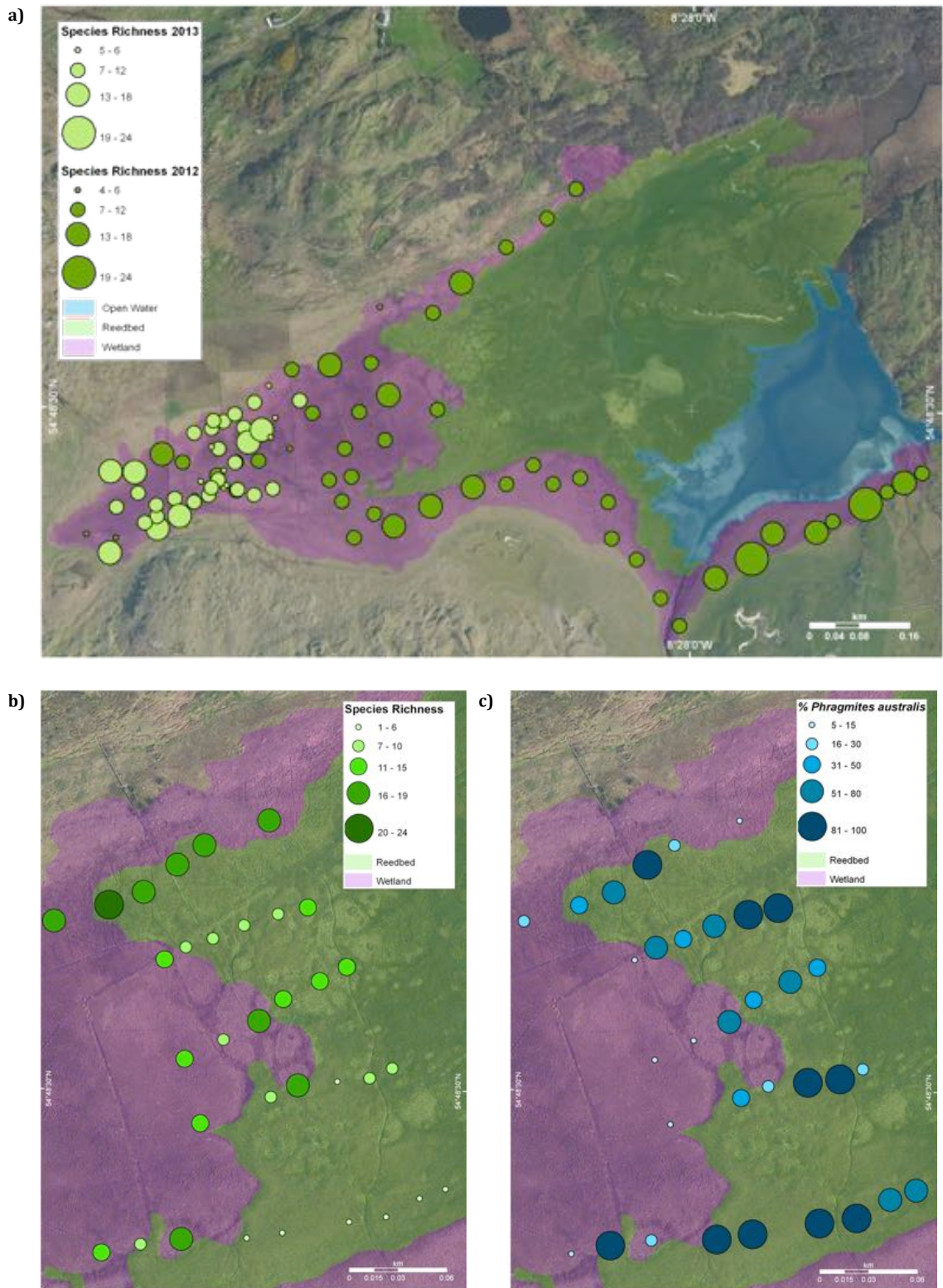


Figure 4.3 Distribution of a) wetland macrophyte species richness, b) reedbed macrophyte species richness, and c) percent cover of *Phragmites australis* at each sample site identified during the June 2012/2013 surveys and the July 2014 survey, respectively.



Figure 4.4 Expansion of the *Phragmites australis* reedbed from 2012 to 2014. The 2012 fringe was delineated from winter 2011- 2012 satellite imagery and a GPS survey was undertaken in July 2014 (Bing, 2012).

Detrended Correspondence Analysis (DCA) of species data reveals relationships between wetland and reedbed macrophyte species and the elevation range of each sample (Figure 4.5). Prior to conducting multivariate analysis on the wetland and reedbed datasets, correlation matrices were used to identify associations between species and samples. Linear and multiple regression analyses were used to assess the significance of these relationships. A significance level of $p < 0.05$ was used for all analyses. Species data were square-root transformed to reduce the effect of zero values before multivariate analysis was undertaken. The data, comprising all species and samples, were then subjected to ordination using CANOCO 4.5. Initially, a Correspondence Analysis (CA) was performed for each dataset; however this resulted in an 'arch effect', which significantly reduces the ability of correctly interpreting the data (Hill & Gauch, 1980; Wartenberg et al, 1987). A DCA was therefore employed as, it not only eliminates this 'arch' effect, the axes in DCA correspond to variability within the dataset and the first axes lengths (wetland = 5.451; reedbed = 4.564) indicate that the data is unimodal (Leps & Smilauer, 2003). Samples were grouped by elevation in order to explore how the species assemblages vary spatially. There are considerable overlaps in species assemblages found in areas of high (5.51-7.50mOD), intermediate (4.01-5.50mOD) and low (3.01-4.00mOD) elevations in the wetland. For instance, at the higher elevations (5.51-7.50mOD) towards the western end of the wetland near the dunes, the water table was noted to be at or above the ground surface within the central area of the wetland. The species associated with this area (e.g. *Briza media*, *Caltha palustris*, *Carex aquatilis*, *Hydrocotyle vulgaris*, *Lychnis flos-cuculi*, *Mentha aquatica*, and *Triglochin palustris*) are characteristic of both drier habitats and wetter conditions. Reedbed samples were also grouped by elevation and the ordination

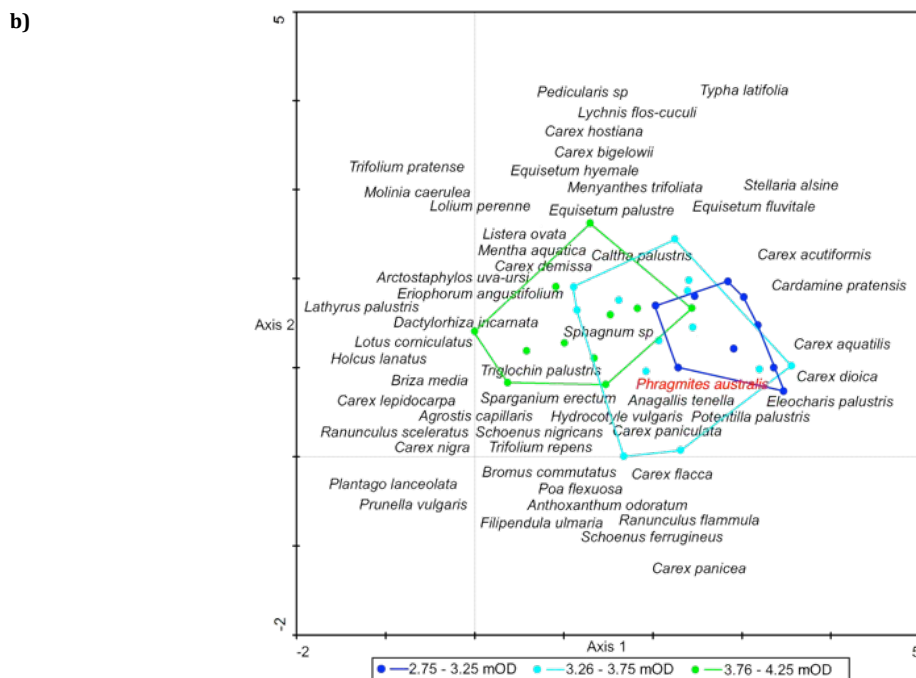
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Figure 4.5 DCAs showing the relationships between the a) wetland macrophytes identified during the June 2012 and 2013 surveys, and b) reedbed macrophytes identified during the July 2014 surveys (*Phragmites australis* is highlighted in red). Samples are grouped based on their elevation (mOD). In a) 29 samples are located at 2.75-3.25mOD; 33 samples are located at 3.26-3.75mOD; and 28 samples are located at 3.76-4.25mOD. Eigenvalues: Axis 1 = 45.6%; Axis 2 = 17%. In b) 9 samples are located at 3.01-4.00mOD; 13 samples are located at 4.01-5.50mOD; and 11 samples are located at 5.51-7.50mOD. Eigenvalues: Axis 1 = 61.9%; Axis 2 = 15.9%.

4.2.2 Lake macrophytes

The spatial distribution of macrophyte density within the open water habitat of the lake reveals that there is no strong relationship between macrophyte PVI (Percent Volume Infested) and elevation or, in this context, lake water depth ($R^2 = -0.284$; $p = 4.4e^{-5}$) (Figure 4.6). This is because high PVI (30.1-75%) is associated with moderate lake depths (3-3.5mOD), low PVI (0-15%) is associated with the shallow central area of the lake (2.5-3mOD) and moderate PVI is associated with the deepest (1-1.5mOD) areas. In terms of species richness and abundance, the lake is dominated by the low-growing (height 5-10cm) charophyte *Chara aspera*, found in 75% of sample sites (Figure 4.7). The second most frequent species is the emergent or submerged tufted rush, *Juncus bulbosus*, found at a third of sample sites. Other abundant species include the fleshy rosettes of *Isoetes lacustris* and *Littorella uniflora*, which were found at 25% and 18% of sample sites, respectively. The variable-leaved *Potamogeton gramineus* was also found at 18% of sample sites. The rarest species is the emergent macrophyte, *Equisetum fluviatile*, found at only one sample site (0.02%). Relatively infrequent species include the aquatic herbaceous perennial, *Lobelia dortmanna* (found at 6% of sample sites), the submerged macrophyte *Myriophyllum alterniflorum* (12%), the submerged floating-leaved macrophyte *Sparganium angustifolium* (12%), the charophytes *Nitella opaca* and *N. translucens* (both 9%), and the upland coastal moss, *Scorpidium scorpioides* (8%). The emergent invasive reed *Phragmites australis* was found in 12% of samples sites, specifically those located close the lake edge.

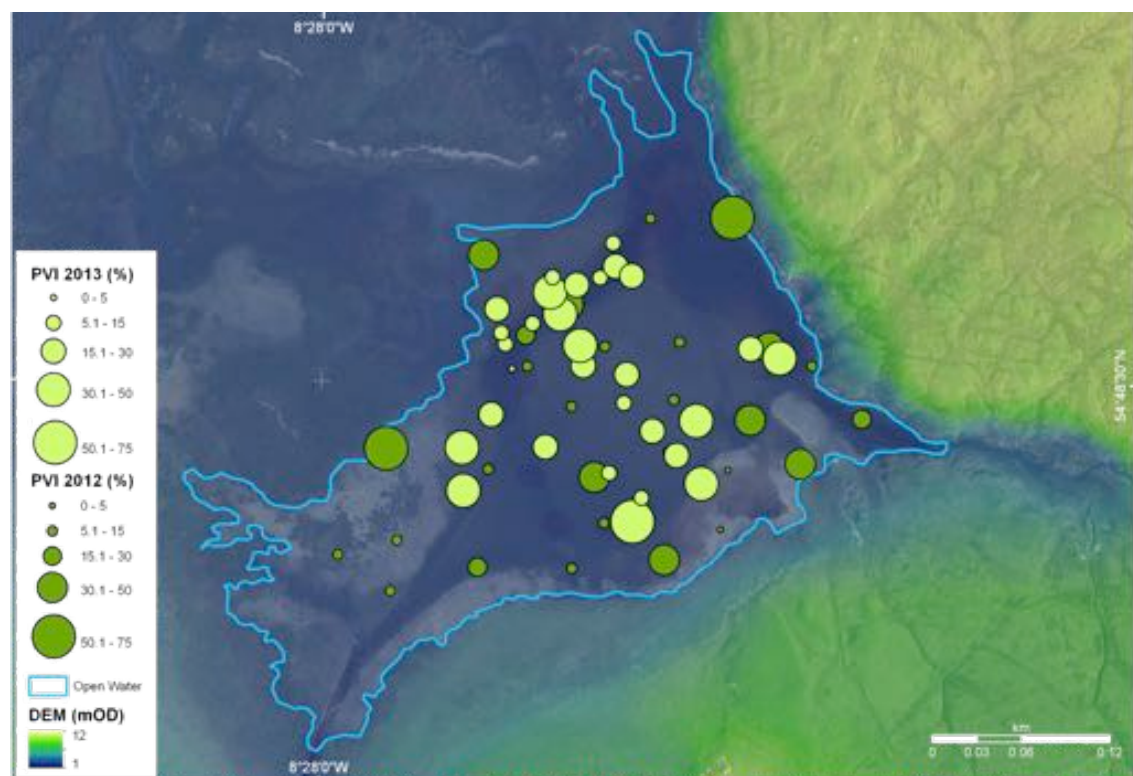


Figure 4.6 Distribution of sample sites across the open water habitat of the lake identified during the June 2012 and 2013 surveys. The percent volume of lake water infested (PVI) with lake macrophytes was recorded at each sample site and is compared to elevation (mOD) via the DEM.

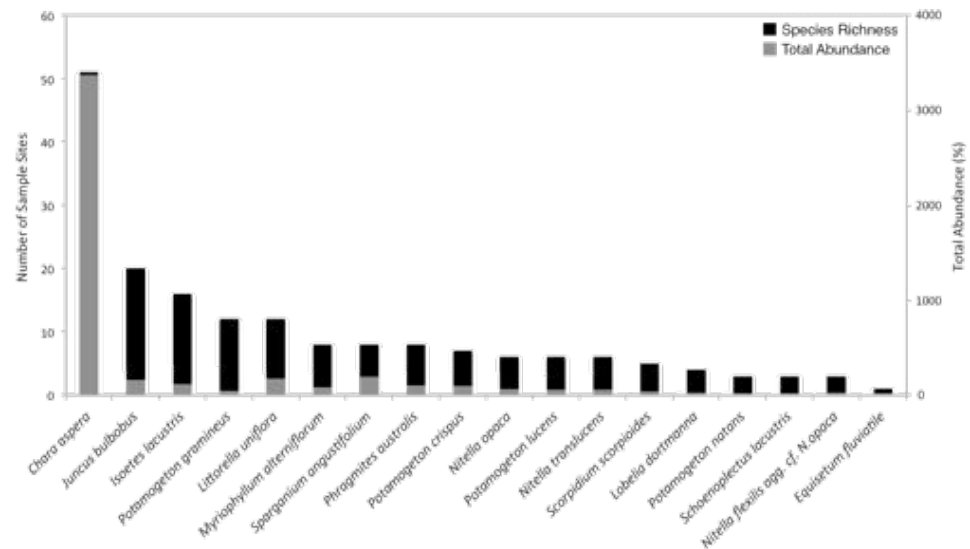


Figure 4.7 Lake macrophyte species richness and total abundance within samples sites in the open water habitat identified during the June 2012 and 2013 surveys.

In accordance with the ordination methods outlined in the previous sections, a DCA was also used to examine the lake species associations, as the 'arch effect' in an initial CA again produced indecipherable results and the DCA first axis length of 4.030 indicates that the data is unimodal (Figure 4.8). There is considerable overlap in the association of species with water depth. For example, species associated with shallow (<0.59m) and intermediate (0.60-0.69m) depths are also associated with deeper areas (>0.70m) of the lake. These include *Chara aspera*, *Isoetes lacustris*, *Nitella flexilis*, *Nitella translucens* and *Scorpidium scorpioides*. Three contrasting species groups are related to the deeper areas of the lake. These include an *Equisetum fluviatile* and *Sparganium angustifolium* pairing, a *Littorella uniflora*, *Lobelia dortmanna*, *Phragmites australis*, *Potamogeton natans* and *Scirpus lacustris* group, and a grouping associated with more moderate depths comprising *Myriophyllum alterniflorum*, *Potamogeton gramineus* and *Potamogeton lucens*.

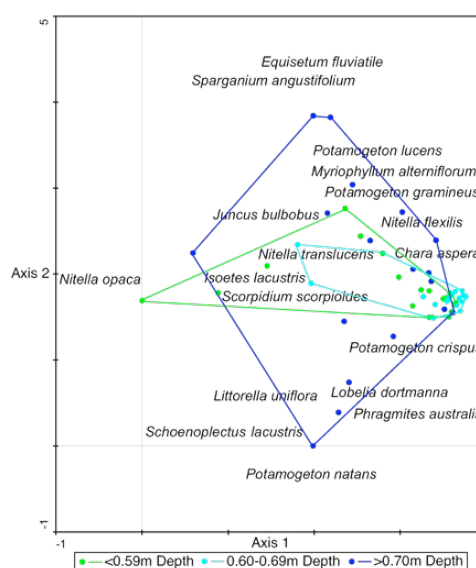


Figure 4.8 DCA showing the relationships between the lake macrophytes species identified during the June 2012 and 2013 surveys. Samples are grouped based on average lake water depth (m) observed during the June 2012 to June 2013 monitoring period. Eigenvalues: Axis 1 = 74.9%; Axis 2 = 29.2%.

The rare aquatic submerged annual macrophyte *Najas flexilis* (Slender Naiad) was found on 12th August 2013 during a survey conducted with the specific intention of relocating the species in Sheskinmore Lough, as it had not been formally identified there since 1981 (Figure 4.9). Only three individual plants were discovered, despite a thorough search in all of the deeper areas of the lake. The three plants that were found were all of good size (20-30cm in height), were growing amongst *Chara aspera*, and were fully-grown; however their complete isolation from one another, and the apparent absence of seedlings indicates the significant rarity of the plant in Sheskinmore Lough at the time of the 2013 survey. In contrast, on 14th July 2014, an abundance of young *Najas flexilis* plants were identified within the lake channel feature, indicating possible inter-annual occurrence within Sheskinmore Lough. It should also be noted that *Najas flexilis* was identified amongst the full spectrum of macrophyte flora and was most commonly located on the soft unconsolidated organic sediments of the shallow channel that meanders from the Duvoge inflow to the southern lake margin.

On 14th July 2014 a transect survey was conducted within the open water habitat of the lake to investigate the distribution of macrophyte species across the channel feature and across the littoral zone. Submerged macrophyte PVI across the channel (digitised from aerial photography) in both the northern and southern transects is greatest (>65%) at the channel base (3.23-3.30mOD; >0.70m depth) (Figure 4.10). The lowest PVI is found on either side of the channel. PVI in the littoral zone of the lake displays a clearer association with elevation. In the southern littoral zone PVI is consistently low (4-24%), except in the higher elevation lake margin (<0.29m depth) where it is slightly higher (25-44%). PVI is similar in the northern littoral zone, with the highest PVI (65-84%) recorded at high elevations (3.31-3.39mOD; <0.39m depth) at the lake margin and the distribution of PVI at lower elevations is more varied, with moderate PVI values (25-64%) recorded at elevations between 3.14mOD to 3.22mOD (<0.59m depth).



Figure 4.9 The locations of *Najas flexilis* (Slender Naiad) plants discovered in August 2013 and the zone of young plants observed along the channel feature within the open water habitat of the lake in July 2014.

The fairly coarse resolution of the DEM shown in Figure 4.10 does not fully represent the detailed topography of the lakebed, which is captured more effectively in Figure 4.11, exposing the subtle changes in species distribution and elevation derived from the 14th July 2014 transect surveys. The transect spanning the deeper northern stretch of the channel (Figure 4.11A) has an asymmetric cross-section with a narrow deep bed, sparsely vegetated steep southern margin and more gradually sloping northern margin. The bed is dominated by *Sparganium angustifolium* interspersed with *Potamogeton crispus*, as well as more sporadically distributed macrophytes including *Myriophyllum alterniflorum*, *Nitella flexilis* and *N. translucens* bounded by the relatively steep channel margin. This section of the channel bed was also covered in *Najas flexilis* seedling plants during the July 2014 survey, but it was also found covering the channel margins, which were dominated by a more varied assemblage of species typical of littoral zones such as *Isoetes lacustris*, *Juncus bulbosus*, *Nitella translucens*, as well as species more characteristic of deeper water including *Potamogeton crispus*, *P. natans* and *Sparganium angustifolium*. In the areas furthest away from the channel, littoral species including *Chara aspera*, *Isoetes lacustris*, *Juncus bulbosus* and *Nitella translucens* dominate, with some sporadic *Potamogeton crispus* plants.

In contrast to the northern section, the southern channel (Figure 4.11B) is more symmetrical but is less well defined, being much shallower and characterised by an undulating bed. The other transects all extend to depths greater than 0.75m depth, however the channel here only extends to a maximum of 0.70cm depth. Like the northern channel, the deepest section is dominated by *Sparganium angustifolium*; however *Nitella flexilis* and *Potamogeton lucens* are more frequent within this stretch of the channel. Less frequent species present at the channel base include *Nitella flexilis*, *N. opaca* and *N. translucens*. Like the northern channel section, *Najas flexilis* seedlings are also rife across the base and sloping sides of the channel. The sloping sides of the southern channel are also characterised by littoral species, specifically *Chara aspera*, *Juncus bulbosus* and *Myriophyllum alterniflorum*. The distribution of macrophyte species also varies across the littoral zone. *Sparganium angustifolium* dominates the deepest northerly end of the southern transect (Figure 4.11C), which is interspersed with *Isoetes lacustris*. *Chara aspera* and *Juncus bulbosus* dominate the shallow areas towards the lake margin, while *Juncus bulbosus*, *Lobelia dortmanna*, *Nitella flexilis*, *Nitella translucens* and *Potamogeton lucens* characterise the steep incline 9-15m from the lake edge. The northeastern transect (Figure 4.11D), which is deepest at its western end, is also dominated by *Sparganium angustifolium* interspersed with *Isoetes lacustris*, however *Juncus bulbosus* and *Chara aspera* are also present at depth. The southern transect differs from the other transects due to its primarily shallow profile. The deepest half of the shallow zone is comprised of *Chara aspera* and *Juncus bulbosus*, which are replaced by stands of *Myriophyllum alterniflorum* and *Potamogeton crispus*, 3.5-5.5m from the lake edge. *Chara aspera* and *Scorpidium scorpioides* dominate the lake margin.

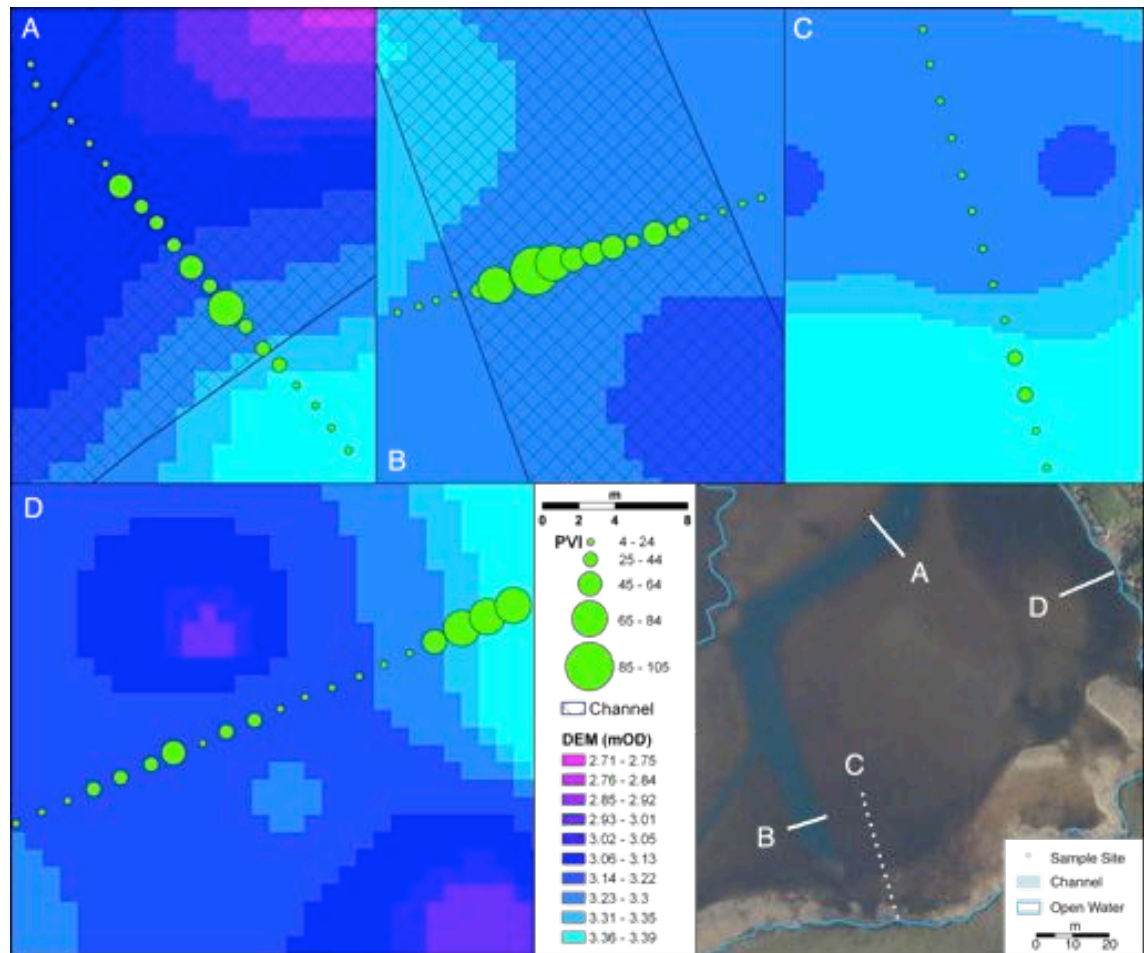


Figure 4.10 Macrophyte PVI within the open water habitat of the lake recorded along four transects located across the 'A' northern and 'B' southern sections of the channel (cross hatching), and two transects extending inwards from the 'C' southern and 'D' northeastern lake edge.

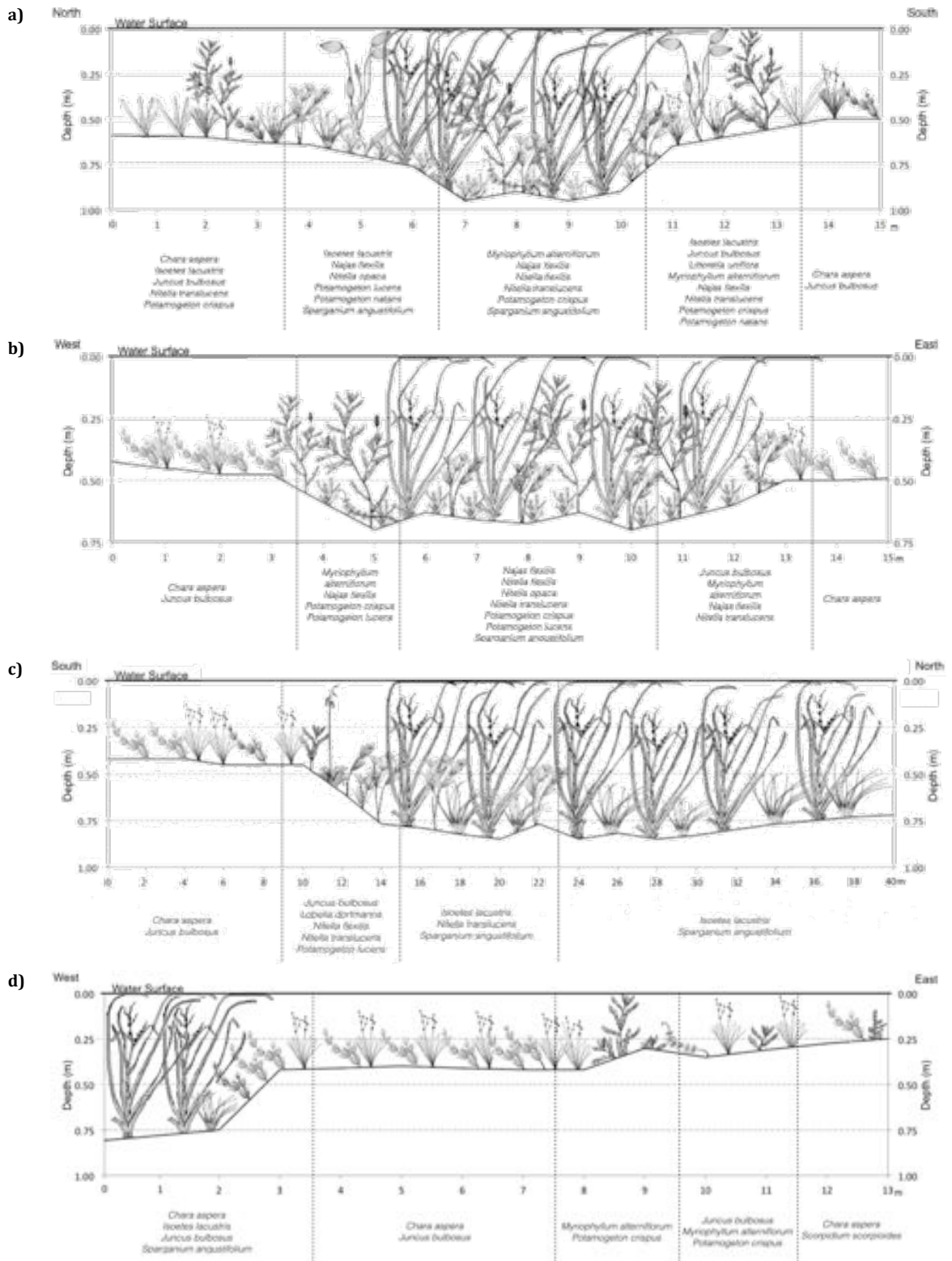


Figure 4.11 Schematic of submerged macrophyte species identified on 14th July 2014 along the two transects across the a) northern and b) southern channel feature and the two transects c) and d) towards the southern and north-eastern margins within the open water habitat of the lake in relation to average water depth (metres below the average water level).

4.2.3 Ecological classification of the lake and wetland system

Aquatic ecosystems contain complex mixtures of flora and fauna, each responding to variations in the physical environment in very different and complex ways. Classification systems seek to artificially compartmentalise environmental features into discrete classes using statistical manipulations and are particularly valuable in aiding and directing conservation management strategies (Gilvear et al, 1994; Allott et al, 2001; Humphries & Meade, 2007; NIEA, 2009; Herrera-Pantoja et al, 2012). For example, these systems produce output classes to distinguish different wetland or lake types; however these outputs vary depending on the nature of the statistic applied and the quality and type of biological data that the statistic is applied to (NIEA, 2009). Classification systems should therefore be used purely as a guide or approximation when categorising ecosystem features and functions. Although two separate ecological surveys were conducted across the wetland and reedbed habitats, only the quadrat placement differed: random sampling across the wetland; and transect sampling across the reedbed. There was considerable overlap in the 53 species recorded (Figure 4.12) so the data were combined for community-based analysis. *Anagallis tenella* was the only species recorded in the reedbed that did not occur in the wetland; 17 species were found only within the wetland

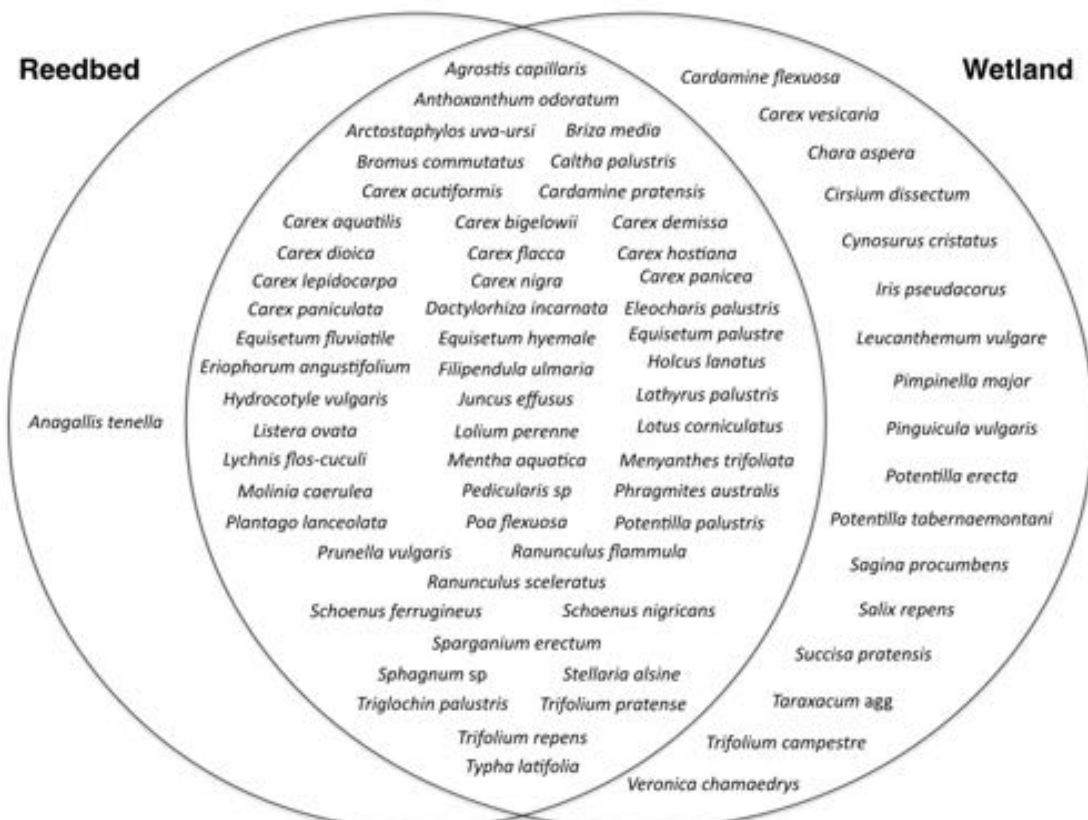


Figure 4.12 Venn diagram detailing species found in the wetland and reedbed habitats, as well as those found in both habitats.

TWINSpan classification of plant data into community types identified six key vegetation community assemblages within the wetland and reedbed habitats (Figure 4.13). TWINSpan is a statistical approach for classifying species and samples, producing an

ordered two-way table of their occurrence (CEH, 2014). The process of classification is hierarchical; samples are successively divided into categories, and species are then divided into categories on the basis of the sample classification. Two divisions and the default TWINSpan cut levels of 0, 2, 5, 10 and 20 best represented the site ecology. The first division in the Sheskinmore wetland-reedbed vegetation analysis split the data into two distinct communities that can be described as *Fen meadow* (comprising species including *Plantago lanceolata*, *Ranunculus sceleratus* and *Trifolium pratense*) and *Peat mire* (comprising *Equisetum fluviatile*, *Cardamine pratensis* and *Sphagnum* sp.). The *Fen meadow* is typical of National Vegetation Classification (NVC) community M22, a lowland fen-meadow community characteristic of moist, base-rich and moderately mesotrophic peats (Table 4.1) (Elkington et al, 2001). *Fen meadow* (community 1) is also indicative of European EUR27 (Natura 2000) machair habitat 21A0. In 2007, the Habitats Committee approved the EUR27 version of the 'Interpretation Manual of European Union Habitats' (EC, 2007). This manual incorporates the 1991 CORINE biotype classification and provides a basis for description of Natura 2000 habitat types from Annex I of the Habitat's Directive so that it can be applied to protected sites such as the wetland and reedbed at Sheskinmore Lough. See Appendix A. for a table containing full NVC and EUR27 community descriptions. Machair-type habitats are complex, resulting from occasional grazing and unintensified rotational cultivation of sandy coastal plains in oceanic environments with cool, moist climates. The wind blown sand contains a significant percentage of shell-derived material, forming a highly calcareous soil of pH >7. Vegetation is herbaceous, with a low frequency of sand-binding species.

In contrast, the second first-level community 2 *Peat mire*, comprises *Equisetum fluviatile*, *Cardamine pratensis* and *Sphagnum* sp., species indicative of wetter conditions associated with lowland mire community characteristic of NVC community M9 containing soft, spongy peats kept permanently moist by moderately base-rich and calcareous waters. *Peat mire* community 2 is indicative of EUR27 transition mire and quaking bog habitat 7140. This is a diverse habitat of peat-forming communities developed at the surface of oligotrophic to slightly mesotrophic waters, characteristic of soligenous and ombrogenous types. The most prominent communities are swaying swards, floating carpets or quaking mires formed by medium-sized or small sedges associated with *Sphagnum* sp. and often accompanied by aquatic communities.

Second-level community 1a *Groundwater mire*, focuses on the presence of *Carex dioica*, *C. flacca* and *C. nigra*, and is typical of NVC M10, groundwater-fed soligenous mire of mineral soils and soft, spongy, shallow peats kept very wet by base-rich, calcareous and oligotrophic waters. This community is indicative of EUR27 blanket bog habitat 7130. It is an extensive bog habitat of flat or sloping ground with saturated soils due to poor surface drainage in oceanic climates with heavy rainfall. These habitats are mostly ombrotrophic, however some lateral water flow may occur. *Sphagnum* sp. plays an important role; however the cyperaceous component is greater and a significant area of vegetation is normally peat-forming. Second-level community 1b *Sedge mire*, comprises *Molinia caerulea*, *Phragmites australis* and *Schoenus nigricans* and is typical of NVC M13, a lowland mire community confined to peat or mineral soils irrigated by moist, base-rich, highly

calcareous, and oligotrophic to slightly mesotrophic waters. It is often found below springs and seepage lines. *Sedge mire* community 1b is also indicative of EUR27 *Molinia* meadow habitat on calcareous, peaty or clayey silt-laden soils 6410. This is a habitat of lowlands to montane levels, on relatively wet nutrient-poor, slightly acidic to calcareous soils with a fluctuating water table, often becoming dry in summer. This habitat stems from extensive management, either due to mowing late in the year or because of peat bog draining.

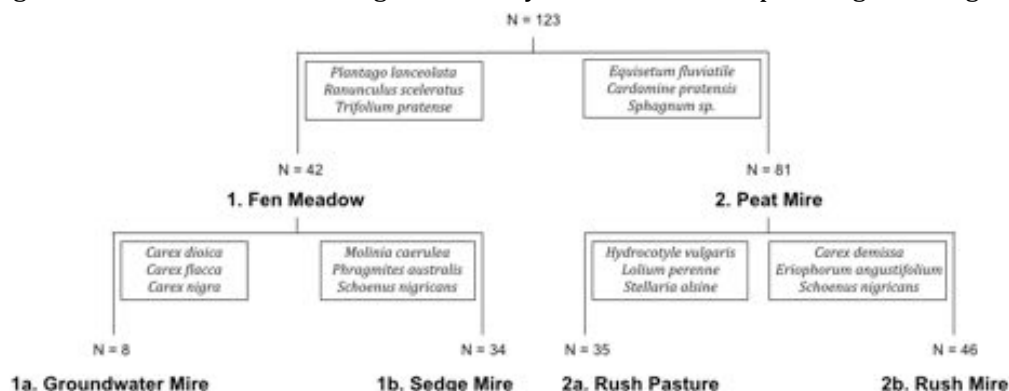


Figure 4.13 Dendrogram TWINSpan result for wetland and reedbed plant species data identified during the June 2012, 2013 and July 2014 surveys. Plant species in boxes represent indicator species for the point of division at which they occurred. First-level communities (1 and 2) and second-level communities (1a, 1b, 2a and 2b) are numbered and 'N' is the number of sample sites associated with each division.

Table 4.1 Wetland and reedbed plant communities identified during the June 2012, 2013 and July 2014 surveys assessed through TWINSpan, their associated species and National Vegetation Classification (NVC) and Natura 2000 'EUR27' codes (Elkington et al, 2001; EC, 2007). See Appendix A. for full descriptions.

Community	NVC	EUR27
1. Fen Meadow	M22	21A0
2. Peat Mire	M9	7140
1a. Groundwater Mire	M10	7130
1b. Sedge Mire	M13	6410
2a. Rush Pasture	M28	7230
2b. Rush Mire	M23	3130

Second-level communities 2a and 2b are formed of *Hydrocotyle vulgaris*, *Lolium perenne*, *Stellaria alsine* species and *Carex demissa*, *Eriophorum angustifolium*, *Schoenus nigricans* species, respectively. Second-level community 2a *Rush pasture* is characteristic of NVC M28, an oceanic lowland rush-pasture characteristic of moist, slightly acidic to neutral, peaty and mineral soils in cool and wet climes. *Rush pasture* community 2a is also indicative of EUR27 alkaline fen habitat 7230. This is a wetland habitat occupied predominantly by peat-producing grass and rush communities developed on soils permanently waterlogged, with a soligenous or topogenous base-rich, often calcareous water supply with the water table at, above or below the surface. This habitat may form part of the fen or mire system, with communities related to transition mires, wet grasslands or spring communities developing in depressions or dune slack systems. These habitats are among those that have undergone the most serious decline across the British Isles, with most having become extinct or endangered (Elkington et al, 2001). In contrast, rush-dominated second-level community 2b *Rush mire*, is located along groundwater

seepage lines, in open water transitions around lakes and beside springs and flushes is indicative of NVC M23, a lowland mire community confined to peat or mineral soils strongly irrigated by base-rich, highly calcareous, and oligotrophic waters. *Rush mire* community 2b is also indicative of EUR27 oligotrophic to mesotrophic standing water habitat 3130. This is an aquatic habitat of short perennial vegetation along the fringes of oligotrophic to slightly mesotrophic lakes. The short annual vegetation is characteristic of pioneer land interface zones around lakes with nutrient poor soils that can experience periodic drying. This habitat is also characteristic of wet dune slacks in the Atlantic region.

The distribution of the TWINSpan communities across the wetland and reedbed is shown in Figure 4.14. First-level community 1 *Fen meadow* is found throughout the wetland system, however it is less prevalent towards the western end. In the reedbed it is almost exclusively associated with the more established vegetation towards the centre of the habitat area. Community 2 *Peat mire*, on the other hand, is more restricted in distribution. This community is located primarily at the western half of the wetland, but is absent from the terminal end. There are also discrete occurrences around the southern and eastern side of the lake. In the reedbed it is almost exclusively confined to the wetland habitat where *Phragmites australis* has encroached. Within the overall distribution of communities 1 and 2, are second-level communities 1a *Groundwater mire*, 1b *Sedge mire*, 2a *Rush pasture* and 2b *Rush mire*. *Groundwater mire* community 1a is the least abundant community, located within a small area at the edges of the western arm of the wetland, on the northern wetland margin and the southwest side of the lake. It is also found across the reedbed survey area in both a north-south and west-east direction, especially in the southwest corner of the habitat. *Sedge mire* community 1b is more widespread, however, located around the north and south sides of the lake and along the northern arm of the central wetland area, but is confined to the central part of the reedbed. In contrast, *Rush pasture* community 2a dominates the area just outside the centre of the western wetland arm and at the wetland edge, including some sporadic sites in the eastern wetland and on the southeast side of the lake. In addition it is found around the fringe of the reedbed, associated with areas furthest away from the core reedbed habitat area. *Rush mire* community 2b dominates the main body of the eastern wetland to the west side of the lake and is frequent along the centre of the western arm of the wetland. It is also found around the fringe of the reedbed and is concentrated within the northwest edge of the reedbed.

The communities were also evaluated based on their elevational associations to enable a direct link between the contemporary ecology and hydrology within the wetland system. Figure 4.15 shows the number of TWINSpan-defined wetland and reedbed communities at each of three elevational ranges, which were determined by dividing the full elevational range within the two habitat areas into classes that were weighted to ensure they contained roughly equal sample numbers. The graphs reveal that there is a clear contrast between the elevation associations of Communities 1 (*Fen meadow*) and 2 (*Peat mire*), with the former associated primarily with the lowest elevation class and the latter associated, largely with intermediate to higher elevations. It should be noted though, that a considerable number of both community samples are also found within the other elevation classes. Examination of Communities 1a (*Groundwater mire*) and 1b (*Sedge mire*)

also reveals strong associations with the lowest elevation range. Proportionally, Community 1b has more samples at lower elevations in comparison to the intermediate elevations and Community 1a, although primarily found at the lowest elevations, is associated more with higher elevations. Communities 2a (*Rush pasture*) and 2b (*Rush mire*), in contrast, are associated with intermediate elevations and are found in roughly equal number within the lowest elevation range. Conversely, the *Rush pasture* community 2a is more strongly associated with the highest elevation range.

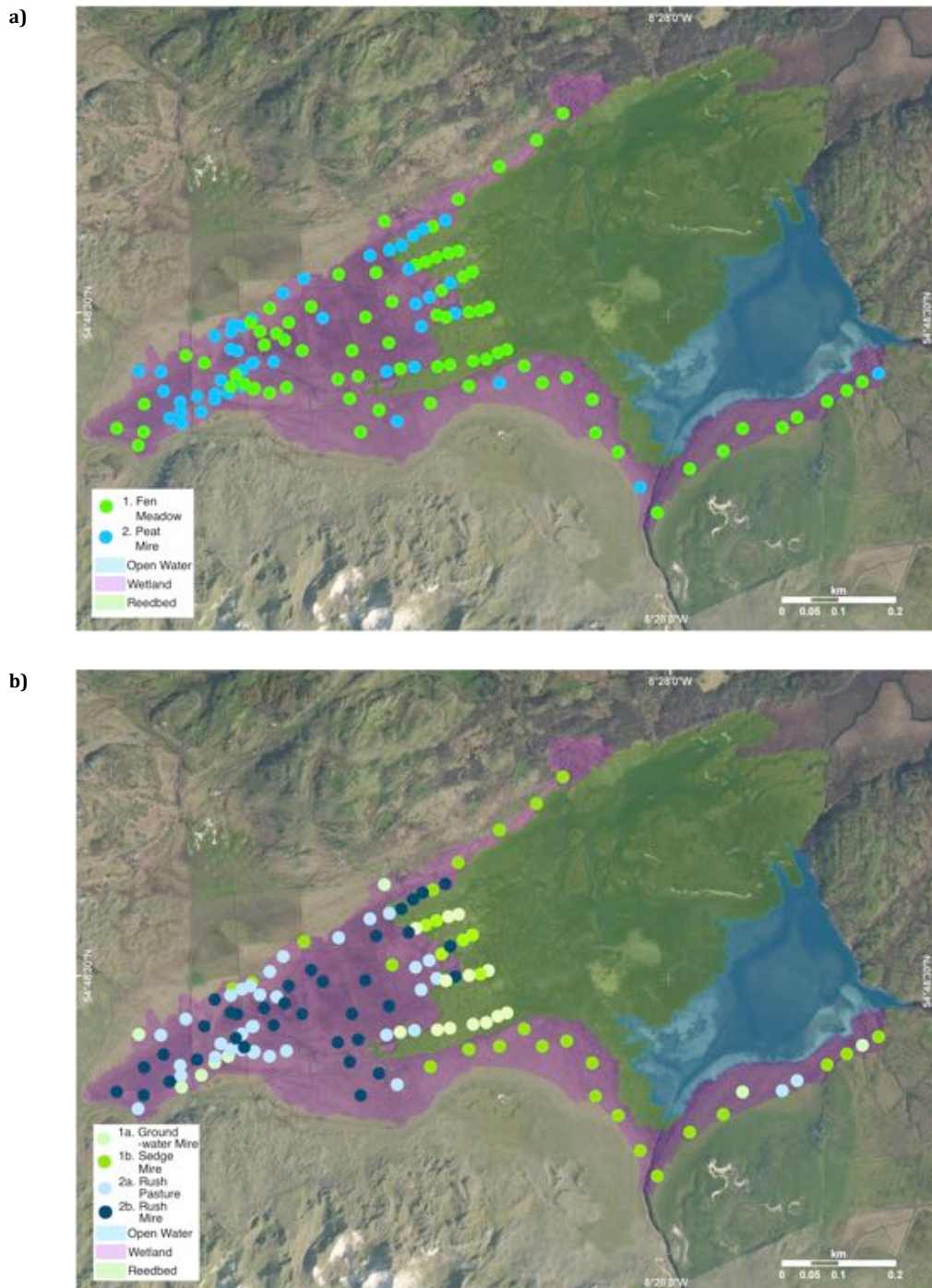


Figure 4.14 The distribution of sample sites corresponding to the TWINSpan-defined communities: a) 1 and 2; and b) 1a, 1b, 2a and 2b; across the wetland and reedbed habitats identified from the June 2012, 2013 and July 2014 survey data.

As well as exploring the distribution and association of the TWINSpan derived communities with elevation it was also important to examine their distribution in the context of local farming and management regimes - specifically grazing and predator control zones (Figure 4.16). During the field surveys it was noted that grazing, by 2/3 horses and up to a dozen cattle, occurs throughout the year in the central area of the wetland arm, with infrequent grazing by two dozen cattle occurring during the winter across all other areas to the west of the lake and outflow. The same herd of two dozen cattle graze the area to the south side of the lake during the summer. Although this will have had no influence on the plant species recorded within the wetland, it is important to note that a predator fence was installed by Birdwatch Ireland in 2014 across the main body of the reedbed and the central wetland area to keep predators such as foxes and stoats away from nesting birds. *Groundwater mire* community 1a is primarily located within the infrequently grazed reedbed margin that now falls within the predator-free zone and displays a minor presence towards the western end of the wetland, falling partly within the regularly grazed area. Conversely, *Sedge mire* community 1b fringes the regularly grazed periphery of the lake and the western end of the reedbed, with a small cluster located within the frequently grazed northern side of the central western wetland arm. The *Rush Pasture* community 2a is also strongly associated with the regularly grazed areas; however it is also fairly widely distributed across the northern periphery of the main arm of the wetland and along the edge of the reedbed. Two samples associated with community 2a are also located within the regularly grazed zone along the south side of the lake. *Rush mire* community 2b is perhaps the most evenly distributed community, located in both frequently and infrequently grazed areas, from the western end of the wetland arm to the northwest and western reedbed margins.

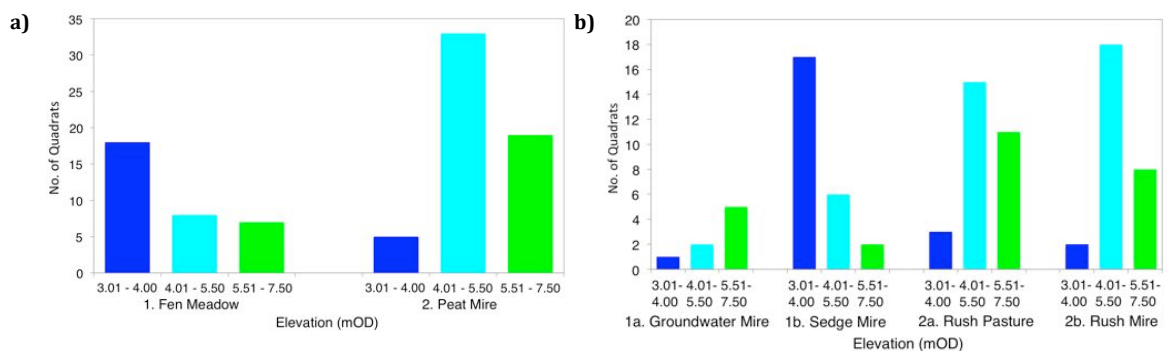


Figure 4.15 Number of TWINSpan-defined wetland and reedbed communities at each of the three elevational ranges used in ordination: a) First-level communities 1. *Fen Meadow* and 2. *Peat Mire*; and Second-level communities b) 1a. *Groundwater Mire*, 1b. *Sedge Mire*, 2a. *Rush Pasture* and 2b. *Rush Mire*.

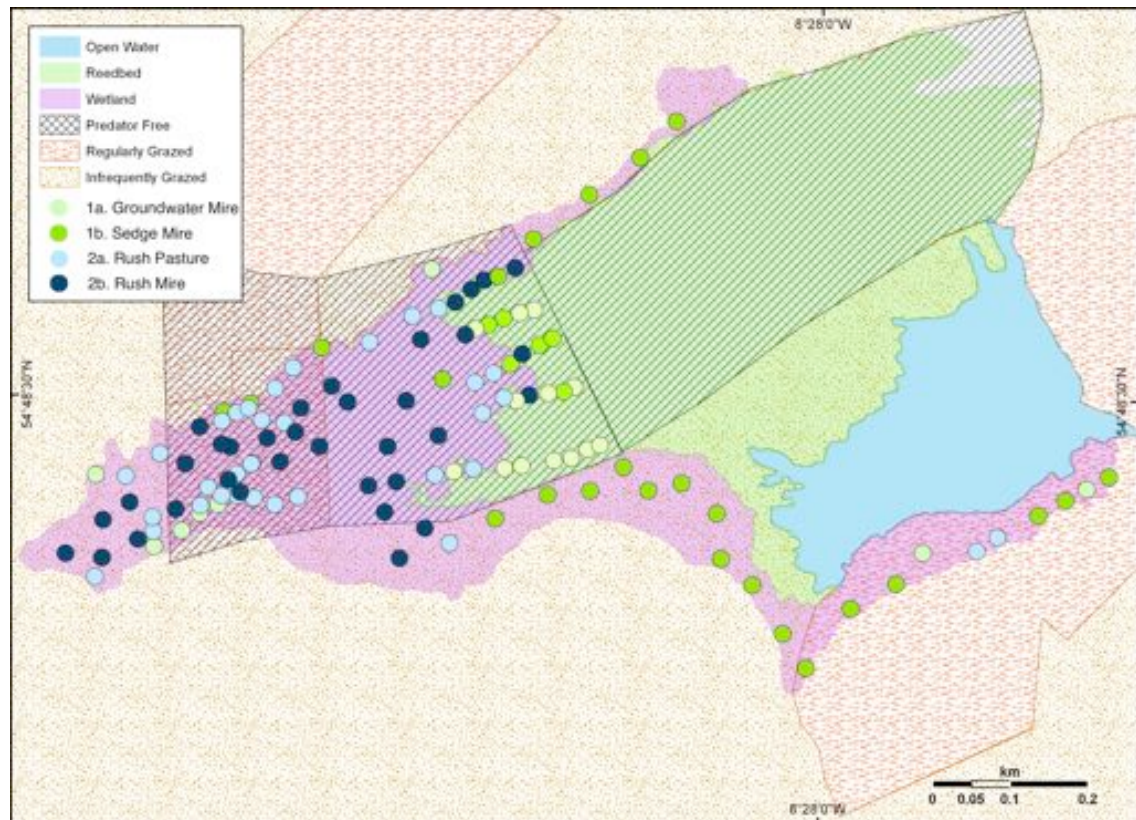


Figure 4.16 Distribution of TWINSpan-defined second-level wetland and reedbed communities 1a, 1b, 2a and 2b in relation to regularly and infrequently grazed areas and predator control zones.

TWINSpan classification of the open water habitat of the lake into community types identified six key vegetation assemblages (Figure 4.17). Several different cut levels and division combinations were applied, however the best representation of the lake ecology occurred using two divisions and the default TWINSpan cut levels of 0, 2, 5, 10 and 20. The first community 1 *Littoral coastal* focuses on species *Juncus bulbosus*, *Nitella translucens* and *Potamogeton gramineus* and is typical of National Vegetation Classification (NVC) lake community E: low altitude, coastal, circumneutral lakes with a high diversity of plant species. See Appendix B. for a table containing full community descriptions. More specifically this community represents littoral regions, comprising high diversities of submerged and floating-leaved macrophyte species indicative of moderate depths (Table 4.2) (Duigan et al, 2006). *Littoral coastal* community 1 is also indicative of Natura 2000 oligotrophic to slightly mesotrophic standing water habitat 3130 (EC, 2007). This is an oligotrophic to slightly mesotrophic lake fringe habitat. Short perennial vegetation of an inner littoral zone exists in close association with a short amphibious annual community that is also present within an outer pioneer zone that survives through growth during periods of drying. The second community 2 *Benthic coastal* is focused on *Chara aspera* and *Littorella uniflora* and is also indicative of NVC lake community E; however *Benthic coastal* community 2 is specific to shallow benthic regions comprising a high diversity of stunted macrophyte species. Community 2 is, however, indicative of Natura 2000 hard oligotrophic to slightly mesotrophic aquatic habitat 3140. This is a habitat characteristic of benthic areas of shallow lakes dominated by charophytes with waters ranging from those

fairly rich in dissolved bases, a pH of 6 to 7 and very clear waters, to those poor in nutrients, base-rich with a pH of often more than 7.5.

In contrast, community 1a *Benthic lowland* is focused on *Isoetes lacustris*, *Nitella opaca* and *Scorpidium scorpioides* and is indicative of NVC lake community C2: slightly acidic, low-altitude lakes, supporting a diversity of plant species. More specifically, however this community is representative of lowland to slightly upland (<250m.a.s.l.), slightly acidic, benthic lake community comprising a high diversity of stunted macrophyte species indicative of shallow areas. *Benthic lowland* community 1a is, however, indicative of Natura 2000 oligotrophic waters with very few sandy minerals habitat 3110. This is a habitat indicative of shallow, base-poor, oligotrophic waters with few minerals. The aquatic to amphibious low perennial vegetation grows on oligotrophic and sometimes peaty soils of the lake banks. Community 1b *Littoral lowland*, however, focuses on *Myriophyllum alterniflorum* and *Potamogeton crispus* and is therefore indicative of a different NVC lake community, in this case I: a lowland, base-rich, littoral lake macrophyte community typical of coastal locations. Conversely, however, *Littoral lowland* community 1b is also indicative of Natura 2000 upstream estuary habitat 1130. More specifically this habitat is the downstream part of a river valley, subject to the extending limit of brackish waters but where there is a substantial freshwater influence.

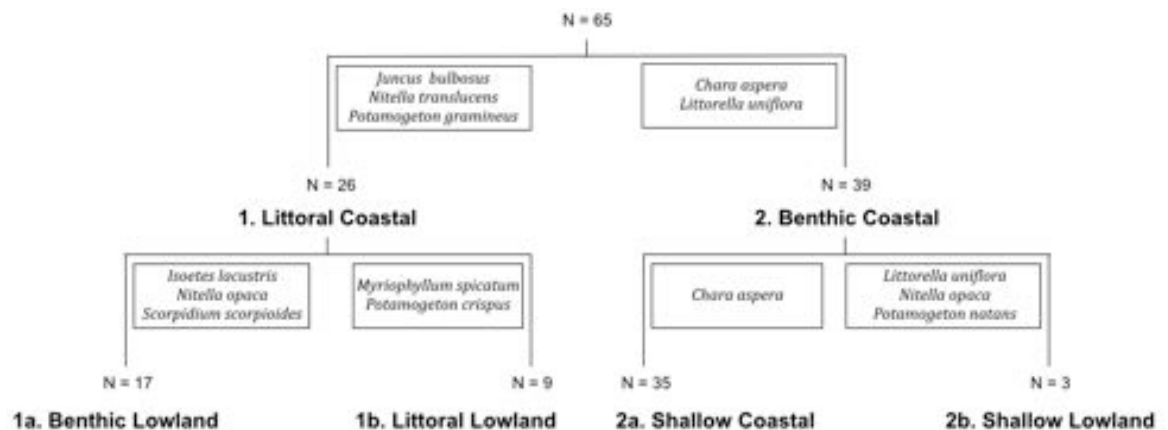


Figure 4.17 Dendrogram TWINSpan result for lake plant species data. Plant species in boxes represent indicator species for the point of division at which they occurred. Communities are numbered and 'N' is the number of sample sites associated with each division. First-level communities (1 and 2) and second-level communities (1a, 1b, 2a and 2b) are numbered and 'N' is the number of sample sites associated with each division.

Table 4.2 Lake plant communities identified during the June 2012 and 2013 surveys assessed through TWINSpan, their associated species and National Vegetation Classification (NVC) and Natura 2000 'EUR27' codes (Elkington et al, 2001; EC, 2007). See Appendix B. for full community descriptions.

Community	Indicator Species	NVC	EUR27
1. Littoral Coastal	<i>Juncus bulbosus</i> , <i>Nitella translucens</i> , <i>Potamogeton gramineus</i>	E	3130
2. Benthic Coastal	<i>Chara aspera</i> , <i>Littorella uniflora</i>	E	3140
1a. Benthic Lowland	<i>Isoetes lacustris</i> , <i>Nitella opaca</i> , <i>Scorpidium scorpioides</i>	C2	3110
1b. Littoral Lowland	<i>Myriophyllum spicatum</i> , <i>Potamogeton crispus</i>	I	1130
2a. Shallow Coastal	<i>Chara aspera</i>	E, I	3140
2b. Shallow Lowland	<i>Littorella uniflora</i> , <i>Nitella opaca</i> , <i>Potamogeton natans</i>	B	3140

Shallow coastal community 2a, focusing solely on *Chara aspera*, is a combination of NVC lake communities E and I. More specifically it is representative of lowland, coastal, circumneutral benthic lake macrophyte communities indicative of shallow and littoral areas. Like community 2, community 2a is also indicative of Natura 2000 habitat 3140, characteristic of benthic areas of shallow lakes dominated by charophytes. *Shallow lowland* community 2b, in contrast focuses on *Littorella uniflora*, *Nitella opaca* and *Potamogeton natans* and is indicative of NVC lake community B: small, low-lying acidic moorland lakes, with a limited range of plant species; however, like communities 2 and 2a, it is also indicative of Natura 2000 habitat '3140'.

The distribution of TWINSpan communities within the open water area of the lake is shown in Figure 4.18. First-level *Littoral coastal* community 1 is located around the lake fringe primarily to the south, southeast and west with a cluster located within the channel feature to the north and one occurrence in the deep area of the northeast corner of the lake. *Benthic coastal* community 2, on the other hand, is concentrated in the central area of the lake and is completely absent from all of the deep areas except for one site where the Abberachrin river enters the lake in the southeast corner. Within the context of communities 1 and 2 are second-level communities 1a, 1b, 2a and 2b. *Benthic lowland* community 1a displays a wide distribution across the lake but is primarily found around the southern side of the lake from the west to the east, as well as a cluster in the central area and within the channel feature in the north. In contrast, *Littoral lowland* community 1b is concentrated within the channel feature but is also located at one site along the northwest edge of the lake and at two sites on the southern side of the lake. *Shallow coastal* community 2a is found across the main body of the lake but is absent from all deep areas. *Shallow lowland* community 2b is the most sparsely distributed community with only three associated sample sites and is also virtually restricted to shallow areas in the north and west, except for one site at the Abberachrin river inflow.

Lake community depth ranges were also examined to provide an ecological context for comparison with contemporary hydrological data. The frequency of each TWINSpan-defined lake community within the three lake depth classes (defined in ordination and converted from elevation to depth) is displayed in Figure 4.19. Due to the flat nature of the lake bathymetry, despite weighting samples to maximise even class coverage, the majority still fell within the shallowest depth range. *Littoral coastal* community 1 is associated primarily with the shallow depths; however it is also significant within the moderate depth class. Similarly, *Benthic coastal* community 2 is strongly associated with the shallowest depths, but is rare in intermediate and deeper zones. Variability is also evident amongst the depth associations of the second tier communities. *Benthic lowland* community 1a is primarily found in shallow depths; however it is also found in moderate numbers at moderate depths. Community 1b, in contrast, is more strongly associated with intermediate to deeper areas. The most dominant second tier community, *Shallow coastal* community 2a, is located almost exclusively in the shallows. *Shallow lowland* community 2b, the least abundant community, is associated with all depths.

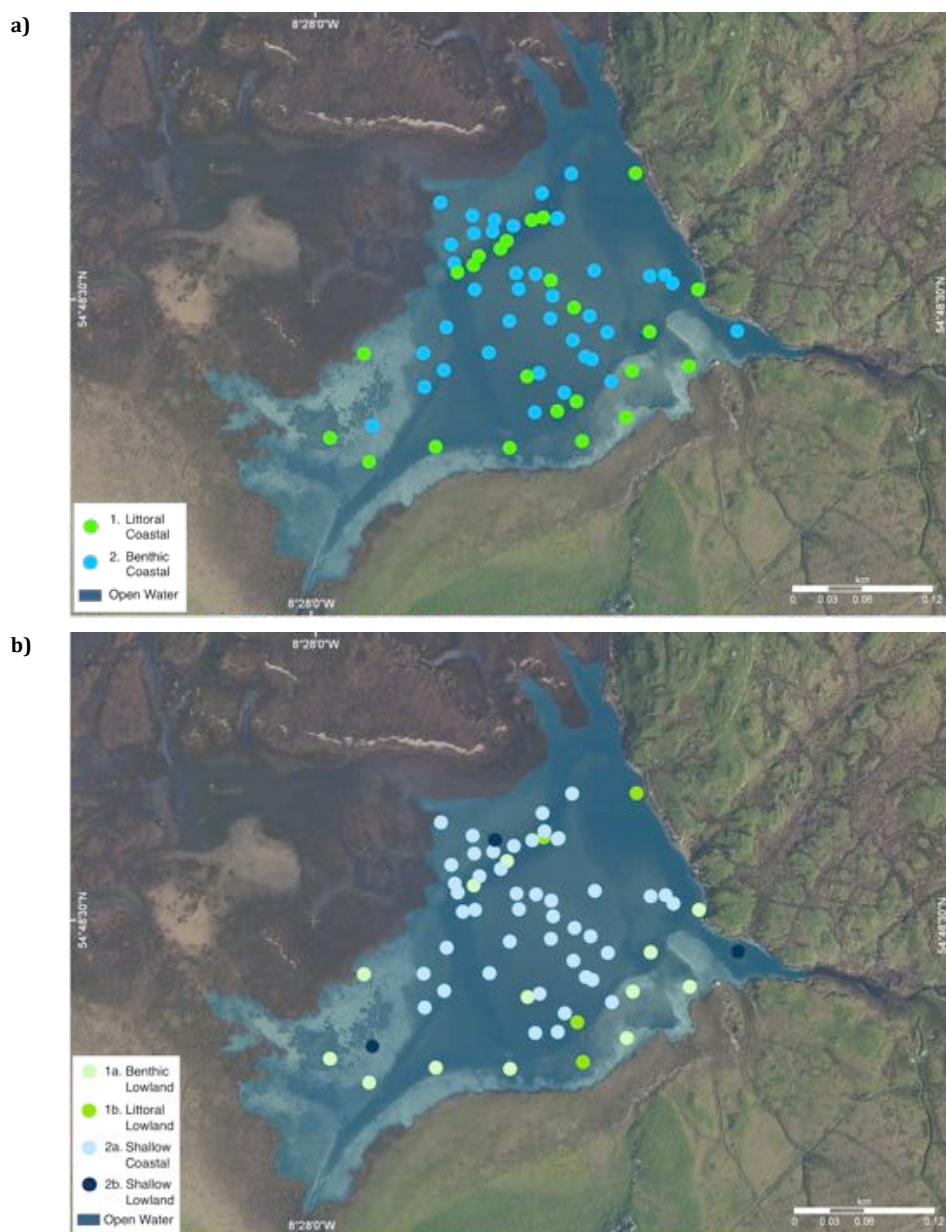


Figure 4.18 GIS showing the distribution of sample sites corresponding to the TWINSpan-defined a) Communities 1 (Littoral Coastal), and 2 (Benthic Coastal), and b) Communities 1a (Benthic Lowland), 1b (Littoral Lowland), 2a (Shallow Coastal) and 2b (Shallow Lowland) across the open water habitat of the lake identified from the June 2012 and 2013 survey data.

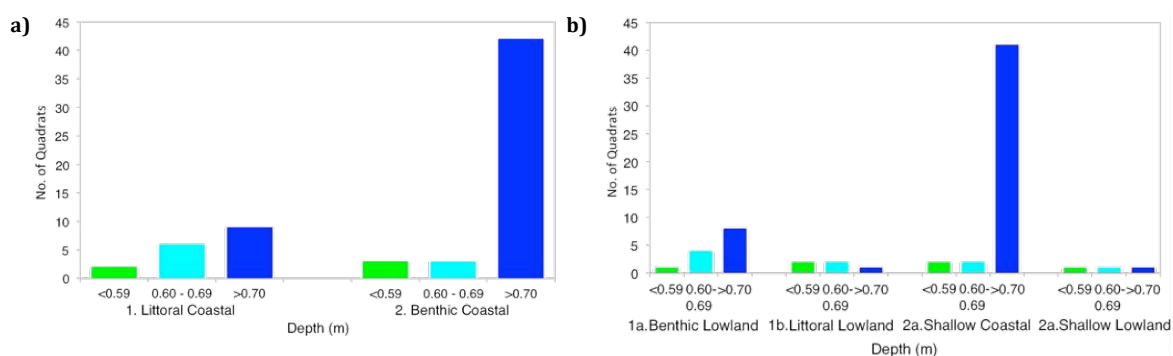


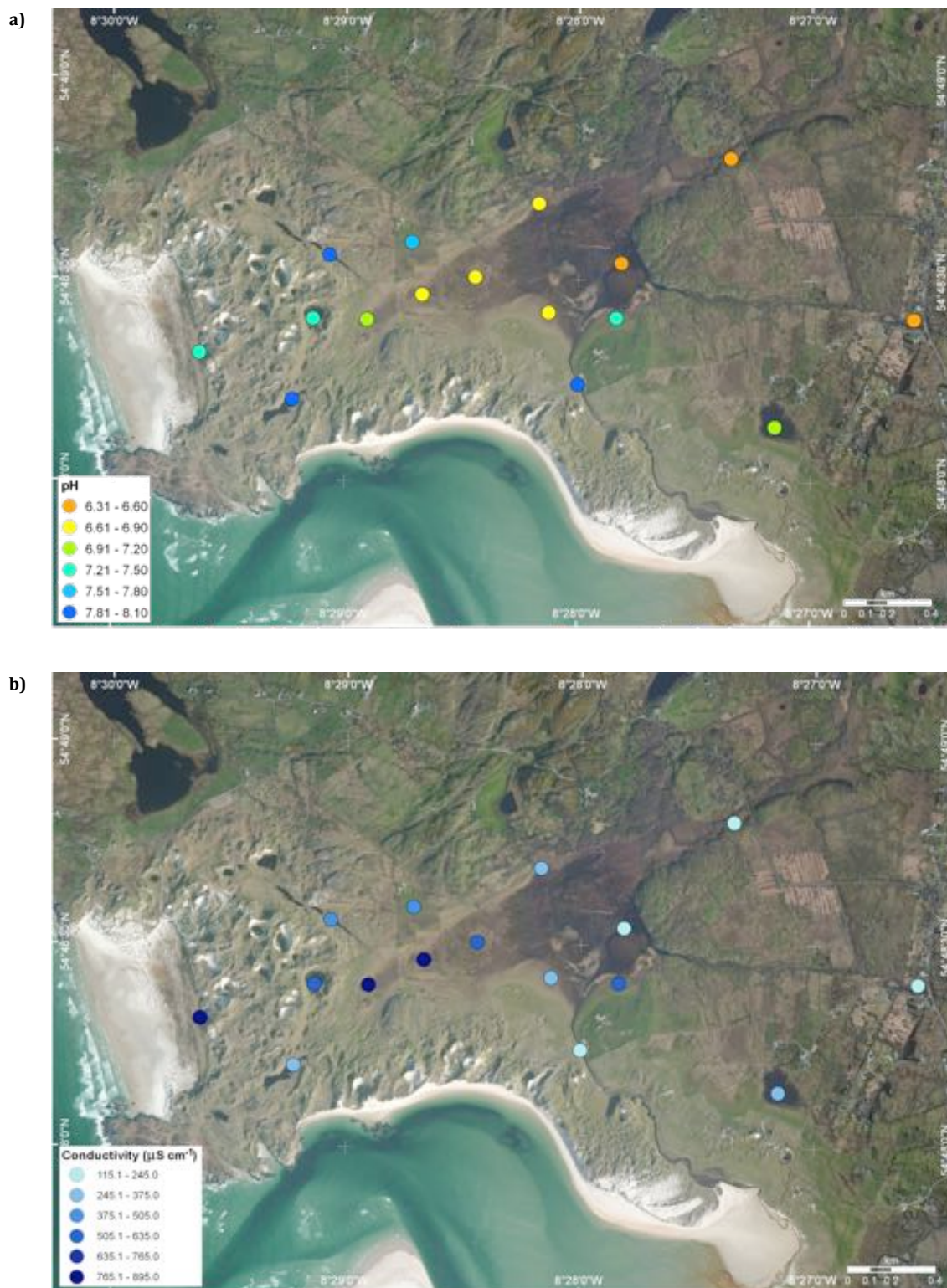
Figure 4.19 Number of TWINSpan-defined lake communities at each of the three depth classes used in ordination: a) First-level communities 1. *Littoral Coastal* and 2. *Benthic Coastal*; and Second-level communities b) 1a. *Benthic Lowland*, 1b. *Littoral Lowland*, 2a. *Shallow Coastal* and 2b. *Shallow Lowland*.

4.3 Edaphic environment

4.3.1 Soil and water chemistry

The distribution of pH across the site reflects the overall sweep of calcareous sand from the southeast over the area of peatland to the northwest (Figure 4.20). The slightly acidic pH of the Duvoige (pH 6.45) and Abberachrin (pH 6.31) rivers provide inflowing water that combines within Sheskinmore Lough to produce a pH within the lake of 6.38. This is in contrast to the neutral pH observed within the smaller Sandfield Lough. pH progressively increases across the wetland in a southwest direction from a slightly acidic pH 6.74 ± 0.03 along the northern and southwest lake margins, to a slightly alkaline pH 7.12 at the far western end of the wetland. Beyond this point, the pH retains this mild alkalinity, with pH 7.24 recorded in the dune blowout. The dune ponds to the north and southwest of the wetland are the most alkaline (pH 7.87 and pH 8.05, respectively). The outflow is also relatively alkaline, with a pH of 7.86. The distribution of conductivity varies in a similar manner to pH. On the whole, the sites with the highest pH levels exhibit moderate to high conductivities, and the sites with the lowest pH, exhibit moderate to low conductivities. The highest conductivities are primarily found towards the southwest of the site, with lower conductivities found towards the northeast, reflecting the pattern of underlying geology. Conductivity is, however generally very low across the site and is only $180 \mu\text{Scm}^{-1}$ within the lake. Exceptions include the far western end of the wetland, which exhibits higher conductivities that are more akin to those recorded in the beach pond ($812 \mu\text{Scm}^{-1}$). In contrast, the dune blowout, which is situated between the wetland and beach, has a lower conductivity ($536 \mu\text{Scm}^{-1}$), indicating a potential hydrological connection between the beach and the wetland. Conductivity within the outflow is low ($205 \mu\text{Scm}^{-1}$), a value that is in accordance with the low conductivity readings (mean $147 \mu\text{Scm}^{-1}$) recorded in the outflow between the sluice and estuary during the week-long hydrological monitoring period in June 2012. These findings confirm that the outflow is unlikely to experience significant prolonged estuarine influence.

The distribution of nutrients is fairly even across the lake and wetland system; however there are some notable variations (Figure 4.21). Low nitrate concentrations are relatively constant ($0.156 \pm 0.002 \text{mgL}^{-1}$) across the western half of the wetland system and into the dunes. The exceptions are the dune blowout and the southern and eastern lake margins, where levels are slightly higher. The lake itself also contains low nitrate levels (0.148mgL^{-1}), with even lower concentrations recorded in the two inflowing rivers ($0.143 \pm 0.002 \text{mgL}^{-1}$). Total phosphate concentrations are also low ($0.01 \pm 0.01 \text{mgL}^{-1}$) across the site. Exceptions are the northern lake margin ($0.04 \pm 0.01 \text{mgL}^{-1}$), the western end of the wetland (0.065mgL^{-1}), and the beach dune pond (0.12mgL^{-1}) where total phosphate concentrations are slightly higher. Similarly, soluble reactive phosphorus concentrations, are also low ($0.004 \pm 0.002 \text{mgL}^{-1}$) across the majority of the site. The highest soluble reactive phosphorus levels ($0.022 \pm 0.002 \text{mgL}^{-1}$) are located within the dune blowout and also to the northwest of the wetland.



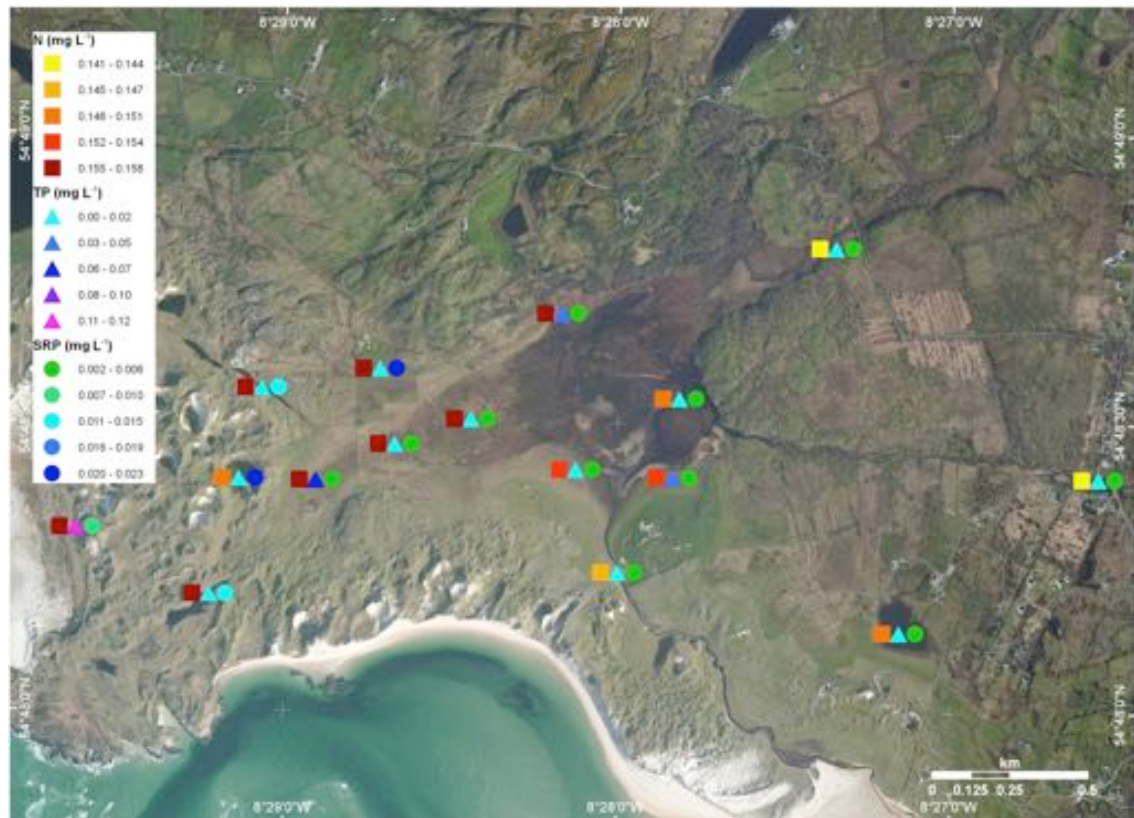


Figure 4.21 Nitrate, total phosphate and soluble reactive phosphorus (mg L⁻¹) measured in-situ in June 2013 at the hydrological monitoring sites located across the lake, wetland and dune systems.

4.4 Hydrology

4.4.1 Temporal scales and patterns in local climatology

Climatological data were collected over the period of two years, from 18th June 2012 to 17th June 2014 at Sheskinmore Lough SPA, and were supplemented with the historical data from the Malin Head meteorological station shown in this section to provide a longer-term context for the hydrological behaviour of the system (ECAD, 2015). Figure 4.22 shows the annual total precipitation over the 1957 to 2014 period (58 years). The year 1999 experienced the highest total precipitation 1339.3mm, with 1967, 1990, 1992, 1998 and 2008 also displaying high total precipitation volumes above 1200mm. The driest years are 1959, 1975, 2001 and 2003, all with less than 900mm total precipitation volumes, the driest of which is 2003 with only 849.8mm. It should be noted that the mean annual total precipitation for the whole data period is 1081.3mm; therefore the survey period for this thesis, from June 2012 to June 2014, is characterised by above-average annual total precipitation with 1094.6mm, 1136.9mm and 1133.9mm, respectively. Statistical analysis comparing the survey data with monthly Malin Head data over the same period reveals that, for the recent survey period at least, precipitation recorded at both sites is statistically similar ($p = 0.002$, $R^2 = 0.832$). Indeed, the survey period is consistent with the long-term trend in annual total precipitation that fluctuates around an average of 1100mm per year from 1978 to 2014. Despite this consistency, significant

variation in total precipitation has also occurred, with five years (13.5% of that 37-year period) displaying the highest total precipitation (1990, 1992, 1998, 1999 and 2008) and six years (16.2% of that 37-year period) with the lowest total precipitation (1983, 1987, 1996, 2001, 2003 and 2010). This is in contrast to the prior 21-year period from 1957 to 1977, where the consistent average varies around 1000mm.

Mean annual temperature data, also obtained from the Malin Head meteorological station, similarly reveal considerable climatological variation throughout the 1957 to 2014 period (Figure 4.22). The majority of years (37 years, 64%) fall within the 9°C to 10°C range; with only a small number of years (1963, 1972, 1979, 1986) falling below 9°C, the coldest being 1979 with an annual mean of 8.63°C. The warmest year was 2007 with an annual mean of 10.8°C. With the exception of 2010, which displayed an annual mean of 9.26°C, all years post-1996 have an annual mean temperature above 9.5°C. The overall trend throughout the whole data period is a gradual increase in the mean annual temperature with time. Nevertheless, a decrease in temperature is evident prior to 1980. The mean for the entire data period is 9.67°C; however when the data are divided into the colder pre-1996 and the warmer post-1996 periods, two distinct means are revealed: 9.46°C and 10.16°C, respectively. Throughout the whole data period there are regular fluctuations in temperature, oscillating between significantly low mean annual temperature troughs at around 9°C and significantly high mean annual temperature peaks of around 10°C. These oscillations range from 7 to 9 years between troughs during the pre-1996 period; however in the post-1996 they are more erratic, with a broader range of temporal reoccurrences from as little as 4 years to as much as 10 years between troughs. The study period, from June 2012 to June 2014, occurs one year after the trough of 2010 and is characterised by below mean annual temperatures in 2012 (9.72°C) and 2013 (9.70°C). 2014, however, is characterised by a mean annual temperature (10.48°C) slightly above the post-1996 mean (10.16°C); however they are still warmer than the pre-1996 mean (9.67°C). Statistical analysis comparing the survey data with monthly Malin Head data over the same period reveals that, for the recent survey period at least, temperature recorded at both sites is statistically similar ($p = <1.1\text{e-}8$, $R^2 = 0.854$).

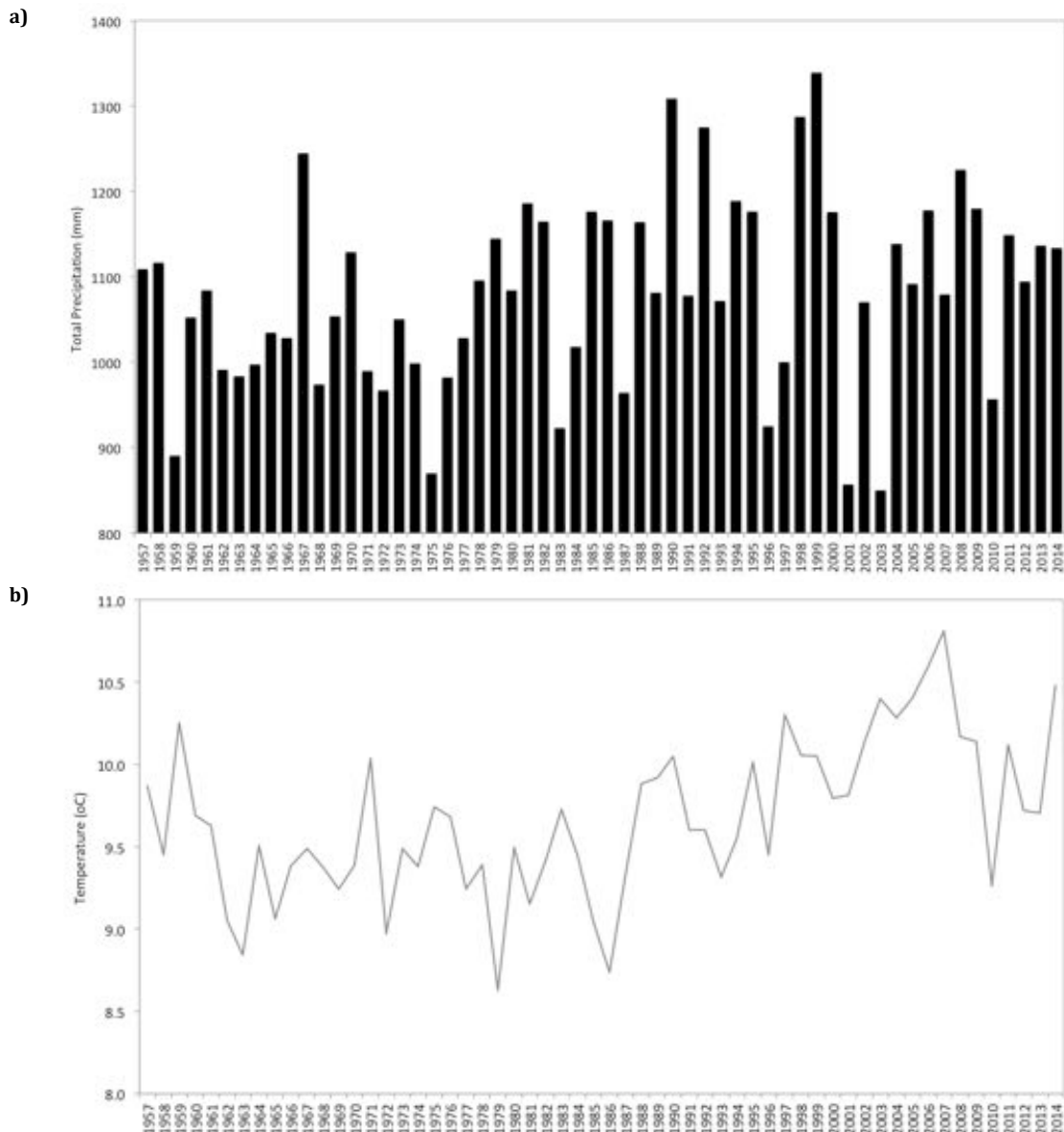


Figure 4.22 Annual total precipitation (mm) and mean annual temperature (°C) from 1957 to 2014 recorded at the Malin Head meteorological station (ECAD, 2015).

Evidence for a seasonal pattern in precipitation is limited, with significant rainfall events occurring throughout the survey period, although the driest periods are confined to the spring and summer months. Precipitation events that exceeded the mean daily precipitation (5.7mm day^{-1}) occur regularly and throughout the year (Figure 4.23, Table 4.3). The exception to this is an extensive 53-day dry (0mm day^{-1}) period that occurred in 2013 from 17th February to 10th April, with a short period of rainfall 2-22nd March 2013 ($<11.3\text{mm day}^{-1}$). There were also three modestly long dry ($<0.4\text{mm day}^{-1}$) periods: 10th-16th July 2012; 19th-26th October 2012; 1st-9th June 2012; 4th July to 9th August 2013; 16th September to 14th October 2013; and 9th-30th April 2014. The total number of wet days ($>0\text{mm}$) was 525 (72% of the year), the wettest day being 18th January 2013 with 56.2mm of precipitation falling in one day. Heavy rain also fell on 28th June 2012 (32.3mm); 2nd October 2012 (34.0mm); 14th December 2012 (34.0mm); 18th January 2013 (56.3mm); 27th January 2013 (34.4mm); 31st January 2013 (36.5mm); 14th December 2013 (36.8mm); and 3rd January 2014 (33.2mm). During the first half of the survey period (from

18th June 2012 to 17th June 2013) these large precipitation events (highlighted in Figure 4.23) occurred over ~3-month intervals and were interspersed by more moderate precipitation events, peaking between 20mm and 30mm. These smaller precipitation events are more frequent than the larger events, occurring from anywhere between 1 day and 3 weeks during the first half of the survey period. In contrast, the second half of the survey period is characterised by a consistently wet period from 13th December 2013 to 21st April 2014, dominated by moderate precipitation events of between 10mm and 20mm, which is preceded and followed by comparatively drier conditions.

Air temperature is strongly seasonal with notable peaks in warmer summer months (June–August 2013 and 2014) and sharp lows in the winter and early spring (Figure 4.23; Table 4.3). The greatest daily temperature variation occurred during winter and early spring, with significant (8°C range) changes between night and day on a sub-weekly basis. The warmest day was 20th July 2013 (mean temperature 22.1°C); the coldest day was 11th March 2013 (mean temperature -0.2°C). ‘Frost’ days, exhibiting close to freezing (0°C) mean daily temperatures, occur from October to November 2012, January to April 2013. The mean temperature for the whole survey period (June 2012–2014) was 10.1°C. During this time, 178 days (48.1%) exhibited temperatures above the overall mean and, on 186 days (50.3%), mean daily temperatures fell below the overall mean. Mean daily temperatures remained above the overall mean throughout the summer months (June, July, August). In contrast, there were many times during the autumn and winter when mean daily temperatures frequently exceeded the overall mean temperature within a number of discrete time bands: 3rd to 13th October 2012; 6th to 20th November 2012; 22nd to 28th December 2013; 2nd to 7th January 2013; 7th September to 9th October 2013; 16th to 28th October 2013; and 8th to 13th December 2013.

The daily total evapotranspiration data recorded during the survey period from June 2012 to June 2014 also reveals a strong seasonal pattern (Figure 4.23, Table 4.3). The mean daily total evapotranspiration for the whole period was 3.5mm. Daily averages peak in the warmer summer months, especially in July and August 2013, with the daily range falling primarily between 5mm and 8mm across the full summer period. Evapotranspiration peaked on 9th June 2013 (9.9mm) during a 6-day period when it consistently exceeded 8mm. Spring evapotranspiration was moderate in 2013 (4mm to 6mm) and moderate-to-high in 2014 (2mm to 4mm). In contrast, the autumn and winter periods were characterised by low evapotranspiration totals, primarily ranging between 0.5mm and 2mm, with noticeable peaks on 28th and 29th December 2012 (4.2mm and 3.8mm, respectively), 31st January 2013 (4.4mm), and from 17th to 20th February 2013 (3.3–4.3mm). The lowest evapotranspiration day occurred on 18th January 2014 (0.2mm).

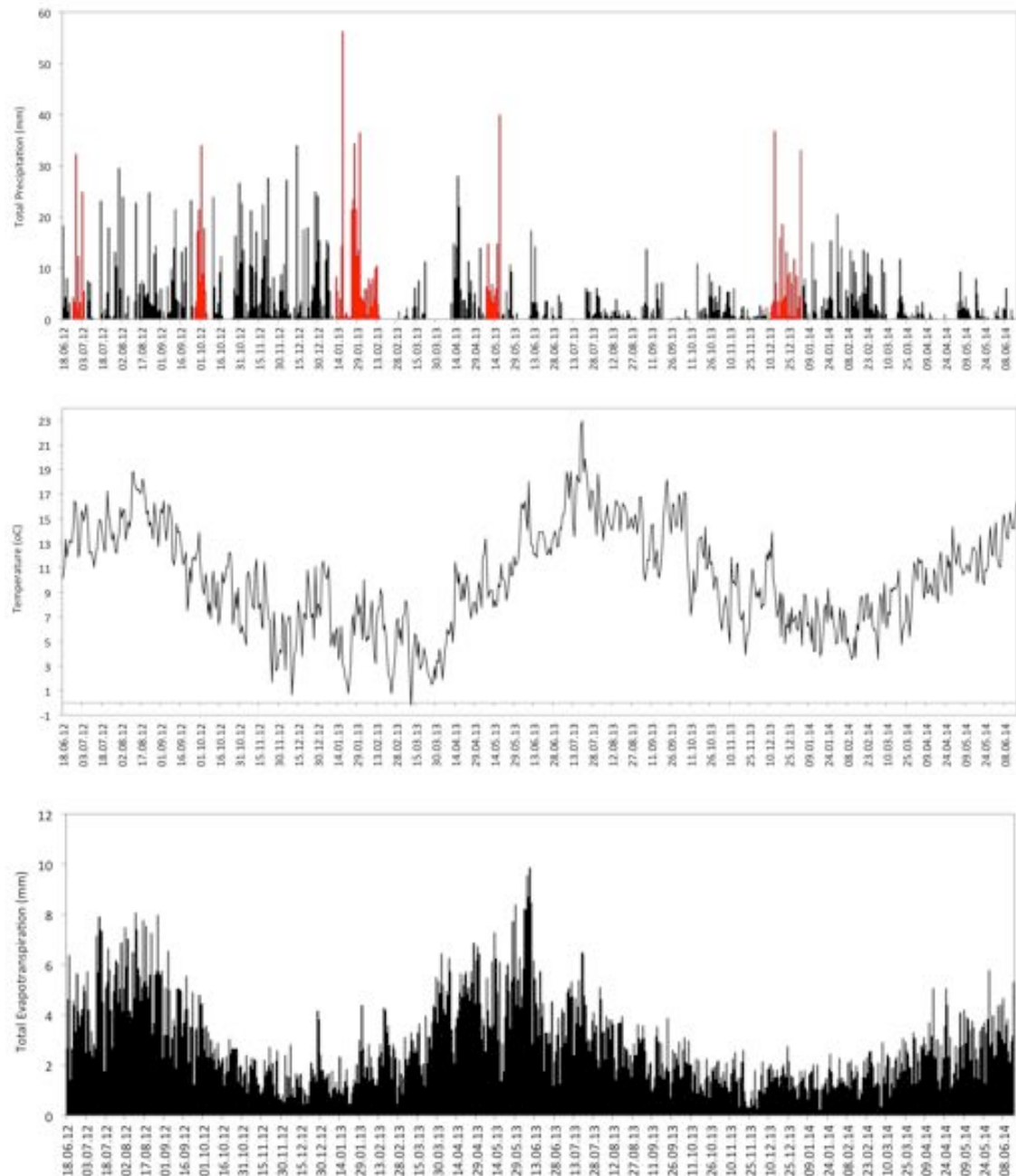


Figure 4.23 Daily total precipitation (mm), mean daily air temperature (°C) and daily total evapotranspiration (mm) over the 2012 to 2014 survey period recorded at the Sheskinmore Lough weather station.

The west Donegal wind climate is temporally variable with no explicit seasonality evident (Figure 4.24, Table 4.3). Mean daily wind speed for the whole survey period was 3.9ms^{-1} . An envelope of variability between 2ms^{-1} and 6ms^{-1} captures most of the week-to-week variation, except in December 2013 when there is a prolonged period of high wind speeds. Wind speeds peak in winter and spring and there are only 4 wind-free days in total, which occur during the turbulent winter and spring period in January, February and April 2014. The highest daily average wind speeds, peaking at over 11ms^{-1} , occur on 28th December 2012, 14th April 2013, 24th December 2013, 3rd and 26th January 2014, and 8th March 2014. The greatest mean wind speeds ($>11\text{ms}^{-1}$) occur predominantly in a southerly direction, with speeds of $5\text{--}8\text{ms}^{-1}$ occurring largely from the southeast to south-southeast. Mean

wind speeds of less than 2ms^{-1} are virtually absent from the southwest quadrant, whilst northeasterly mean wind speeds do not exceed 5ms^{-1} . Gale force winds greater than 17ms^{-1} , occur in every direction except for the east, east-northeast and northeast and occur most frequently from southerly and westerly directions.

Table 4.3 Summary of the key daily and monthly weather characteristics over the 2012 to 2014 survey period recorded at the Sheskinmore Lough weather station.

Weather Feature			
Precipitation	Characteristics	Total Precipitation (mm)	Timescale
	Mean Absolute Daily Precipitation	5.7	730 days
	Mean Absolute Monthly Precipitation	158.8	24 months
	No. of Wet Days	>0	525 days
	No. of Dry Days	>0	205 days
	Wettest Day	56.2	18 th January 2013
	Wettest Month	312.1	January 2013
	Driest Month	13.2	April 2014
Temperature	Characteristics	Temperature (°C)	Timescale
	Mean Daily Temperature	10.1	730 days
	Mean Monthly Temperature	9.6	24 months
	Warmest Day	22.1	20 th July 2013
	Coldest Day	-0.21	11 th March 2013
	Warmest Month	15.8	August 2012
	Coldest Month	4.06	March 2013
Evapotranspiration	Characteristics	Evapotranspiration Rate	Timescale
	Mean Absolute Daily Evapotranspiration	3.5 mm day^{-1}	730 days
	Mean Absolute Monthly Evapotranspiration	$77.9\text{ mm month}^{-1}$	24 months
	Highest Evapotranspiration Day	9.9 mm day^{-1}	9 th June 2013
	Lowest Evapotranspiration Day	0.2 mm day^{-1}	18 th January 2014
	Highest Evapotranspiration Month	$174.7\text{ mm month}^{-1}$	August 2012
	Lowest Evapotranspiration Month	$32.4\text{ mm month}^{-1}$	January 2014
Wind Speed	Characteristics	Wind Speed	Timescale
	Mean Daily Wind Speed	3.9 ms^{-1}	730 days
	Mean Monthly Wind Speed	4 ms^{-1}	24 months
	Highest Wind Speed Day	$11.4\text{ ms}^{-1}\text{day}^{-1}$	8 th March 2014
	Highest Wind Speed Month	$7.3\text{ ms}^{-1}\text{month}^{-1}$	December 2013
	Lowest Wind Speed Month	$2.3\text{ ms}^{-1}/\text{month}$	October 2012

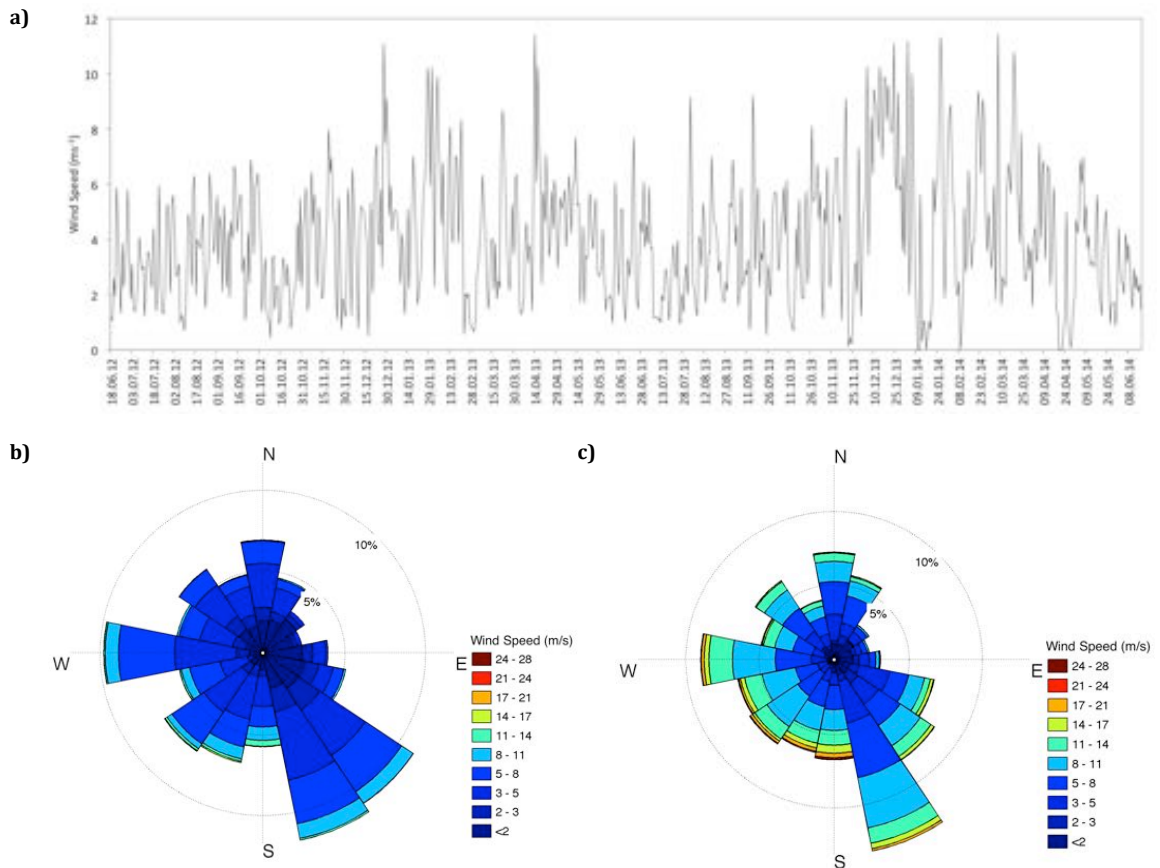


Figure 4.24 a) Daily average wind speed ms^{-1} over the 2012 to 2014 survey period recorded at the Sheskinmore Lough weather station. Wind rose plots showing the b) mean wind speed (m/s) and c) highest wind speeds (m/s) based on the Beaufort Scale with wind direction recorded during the 10-minute period at each 30-minute recording interval during the June 2012 to June 2014 survey period.

The precipitation patterns and trends identified on a daily basis are also reflected in the mean monthly totals (Figure 4.25, Table 4.3). There is no clear seasonal structure of precipitation, with very dry months following very wet months (e.g. December and January to February and March). April 2014 was the driest month with only 13.2mm. The wettest month was January 2013 with 312.1mm. The mean absolute monthly precipitation for the whole survey period was $121.8\text{mm month}^{-1}$. In contrast, the mean monthly temperature and evapotranspiration exemplify the strong seasonal patterns observed from the daily data. The warmest months were July (15.5°C) and August (15.3°C). Monthly temperatures fell throughout the autumn and winter to a low in the spring during the months of February (5.8°C) and March (5.9°C). Evapotranspiration was highest in July (117.2mm) and August (122.6mm), falling to lows in November (39.6mm), December (42.3mm) and January (32.4mm). Mean absolute evapotranspiration for the whole survey period was 77.9mm month^{-1} . Like precipitation, monthly average wind speed fluctuated during the survey period, displaying no clear seasonal pattern. The winter months exhibited the highest wind speeds, especially in December (5.9ms^{-1}), and the lowest monthly wind speeds were recorded in June (3.17ms^{-1}) and July (2.77ms^{-1}). The mean monthly average wind speed for the whole survey period was $4\text{ms}^{-1}\text{month}^{-1}$.

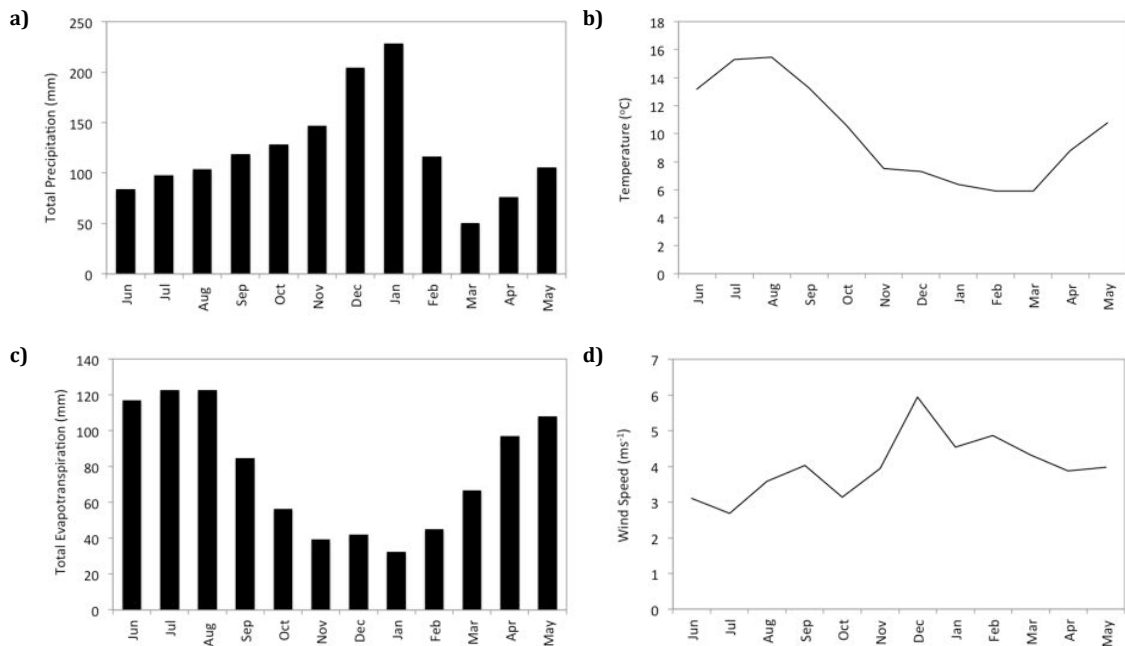


Figure 4.25 Mean monthly a) absolute precipitation (mm), b) temperature (°C), c) absolute evapotranspiration (mm), and d) wind speed (ms⁻¹) over the 2012 to 2014 survey period recorded at the Sheskinmore Lough weather station.

4.4.2 Lake and wetland hydrology

Sheskinmore Lough and wetland are notable flat features in the context of the surrounding topography (Figure 4.26). The lake lies at the confluence of the Duvoge and Abberachrin river valleys where the bedrock valley framework broadens, accommodating the Loughros More estuary. The east of the lake-wetland system is bound by bedrock-framed hinterland whereas the west of the site is dominated by undulating, and in places high (summits of 29-39mOD), sand dunes. Several large blowouts within the dune system produce localised depressions at 5-8mOD. In contrast, the wetland habitat ranges from 3.4mOD to 9.5mOD and the open water (lake) habitat from 1.3mOD to 3.5mOD. The lakebed level lies within the upper intertidal elevation range of the estuary and coast (locally). The deep steep-sided Duvoge river and the artificially deepened outflow channel are clearly evident leading into and out of the lake area. The deepest part of the lake is found close to the interfluvium between the Duvoge and Abberachrin rivers, and elsewhere the bathymetry is quite hummocky with fragmented deeper or shallower patches. The gradual rise in elevation across the wetland to the west is relatively featureless in contrast.

The highest habitat elevation ranges within Sheskinmore Lough SPA are dominated by upland habitat (2.7-71.3mOD) with a mean elevation of 25.7mOD and sand dunes (-0.9-40.8mOD) with a mean of 13mOD (Figure 4.27). Peatland and heathland habitats reach maximum elevations of 28.9mOD and 20mOD, respectively. The Atlantic salt meadow contains the lowest elevation of -1.4mOD. The Sheskinmore Lough aquatic habitats are found within a narrower range of elevation: the wetland at 1.3-8mOD, reedbed at 1.3-4.2mOD and lake open water at 1.3-4.3mOD (1.3-3.5mOD for Sheskinmore Lough and 3.5-

4.3mOD for Sandfield Lough). The machair and wet grassland habitats that divide these two open water habitats range in elevation from 1.3mOD to 16.1mOD.

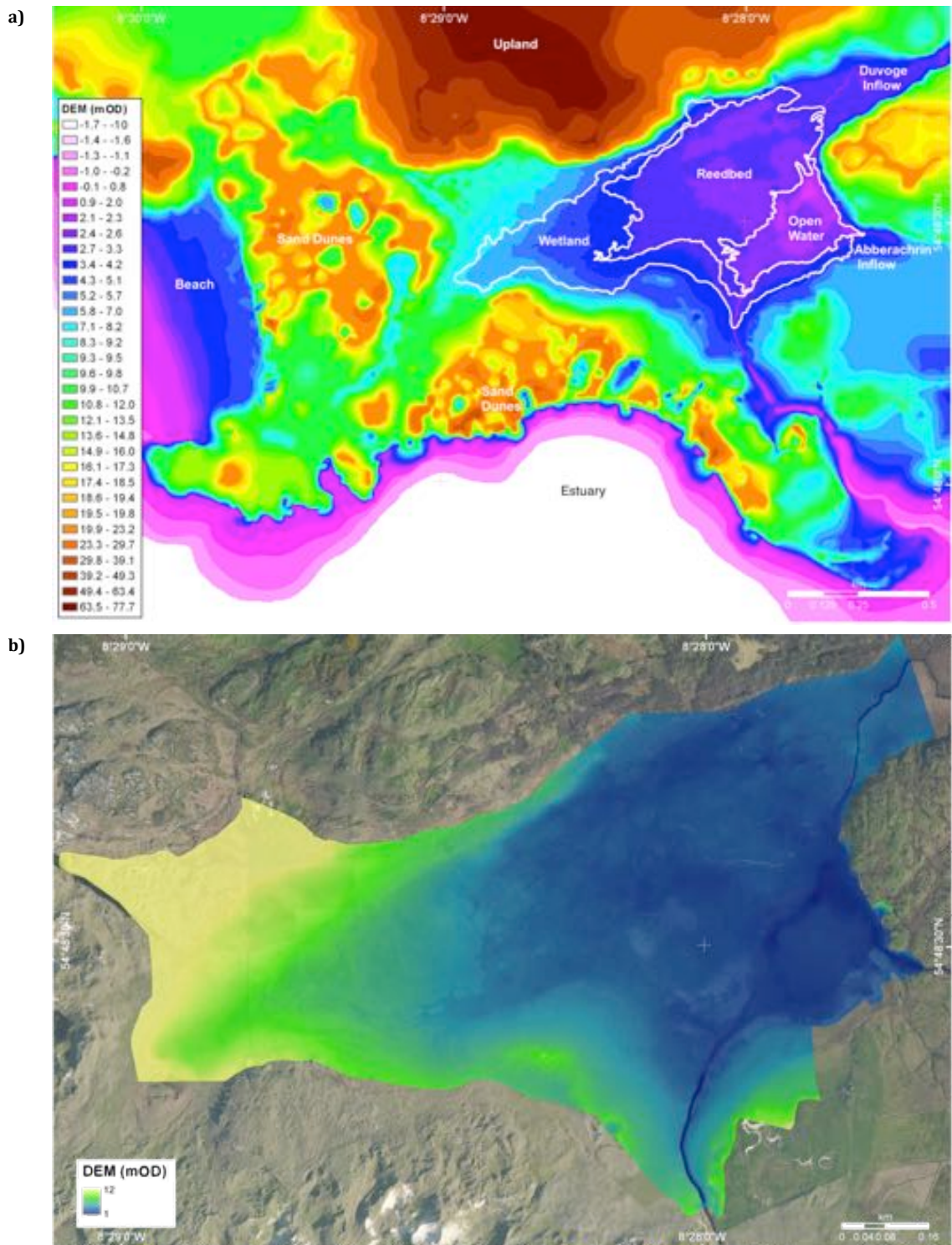


Figure 4.26 Digital elevation models (DEMs) of the study site: a) 1m resolution DEM of the study area based on contour data with habitats labelled and the three key habitat areas outlined (wetland, reedbed and open water area of the lake), and b) 1m resolution DEM of the lake and wetland based on dGPS topographic and bathymetric survey data.

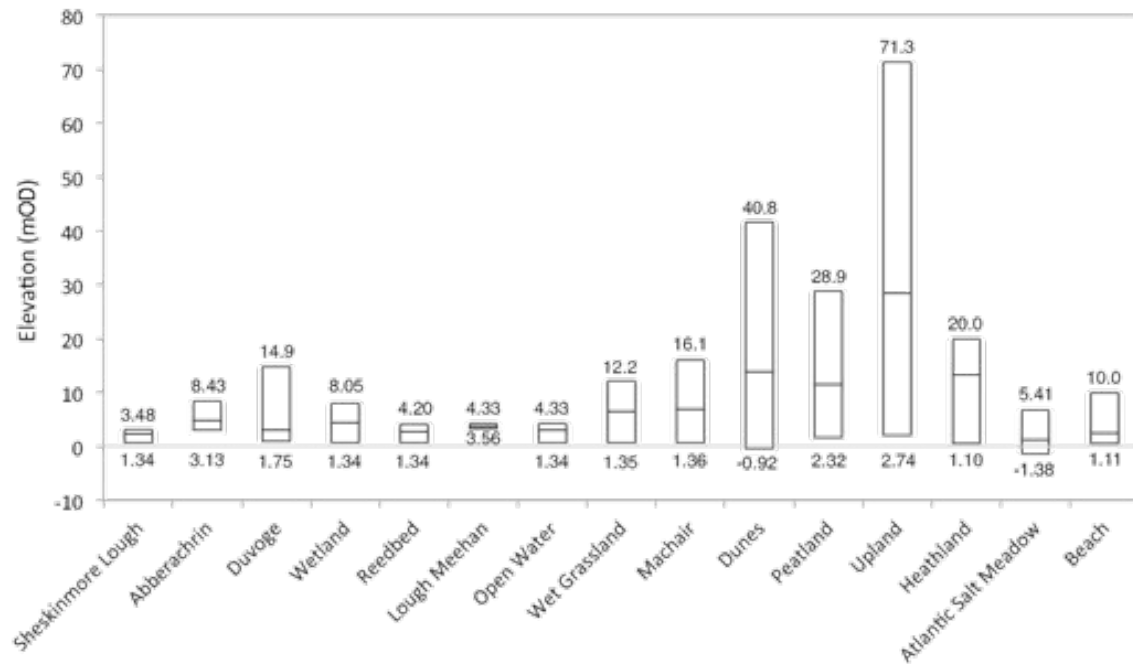


Figure 4.27 Elevation ranges in metres above sea level (mOD) of the habitats at Sheskinmore Lough SPA based on contour data. Note that maximum and minimum elevation values are included within the graph, as is the mean, which is represented by the line inside each bar. Also note the 'Open Water' habitat represents the full range of the Sheskinmore Lough and Sandfield Lough elevation ranges that are also included as separate habitat areas.

The elevation across the hydrological monitoring sites (Figure 4.28) varies greatly, including within the habitat areas of the wetland and dunes. The elevation at the western end of the wetland at W1 is around 4m higher than the monitoring sites by the lake (W5-7). Similarly, the contrast in elevation between the dune blowout (D1) and the beach pond (D2) is considerable at nearly 3m. The elevation of the two lakes is also very different, with Sheskinmore Lough (L3) 2m lower than Sandfield Lough (L6). The monitoring sites with the highest elevations are the dune pond D1 (7.02mOD), the Abberachrin river L2 (6.09mOD) and the base of Mullyvea S3 at 8mOD.

The lowest hydrological monitoring site is the sluice L4 (3.05mOD). The sluice structure comprises a concrete conduit with adjustable gates (Figure 4.29) that when open has a minimum elevation of 3.02mOD, but when closed present a barrier at 4.05mOD. Table 4.4 lists the dates when the sluice was adjusted. The longest period (58 days) when the sluice was open was 10th July to 5th September 2013. The National Parks and Wildlife Service (NPWS) manipulated the sluice, except for one incidence when the sluice was opened by a landowner leading to the shortest (8 days) open sluice period from 19th to 27th October 2012. The longest period (118 days) when the sluice was closed was from 27th October 2012 to 22nd February 2013, and the shortest (13 days) between 17th and 30th May 2013.

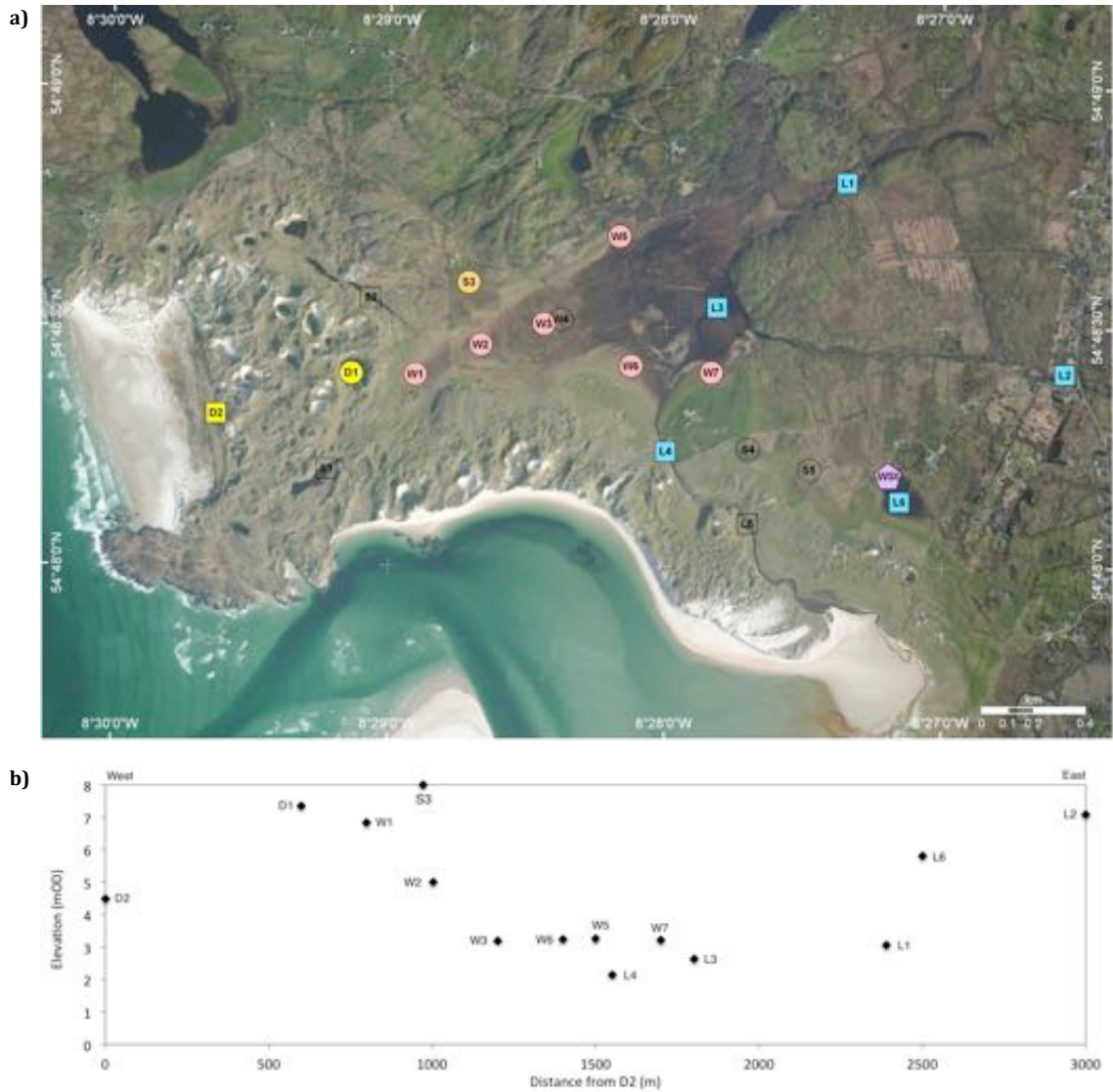


Figure 4.28 a) The location of the on-site weather station (WSt) and the hydrological monitoring sites contributing data to the two-year survey (18th June 2012 to 17th June 2014). The sites include those in the inflows (Duvoge L1; Abberachrin L2), Sheskinmore Lough (L3), the sluice outflow (L4), Sandfield Lough (L6), the wetland system (W1-3; W5-6), the surrounding area (S3), the dunes (D1) and the beach pond system (D2). Note that colourless sites L5, W4, S1, S2, S4 and S5 are not included in this survey. b) Elevational distribution of the hydrological monitoring sites from D2 across the lake and wetland system from west to east for the two-year survey period.



Figure 4.29 The elevations of the sluice gate when it is open and closed relative to the elevation of hydrological monitoring site L4 in the outflow. Note when the sluice is open water passes underneath the gate, an event that occurs below the water level and is therefore not visible in this photo.

Table 4.4 The dates when the sluice was opened and closed during the 730-day hydrological monitoring period from 18th June 2012 to 17th June 2014 (Emer Magee pers. comm. 8th August 2014).

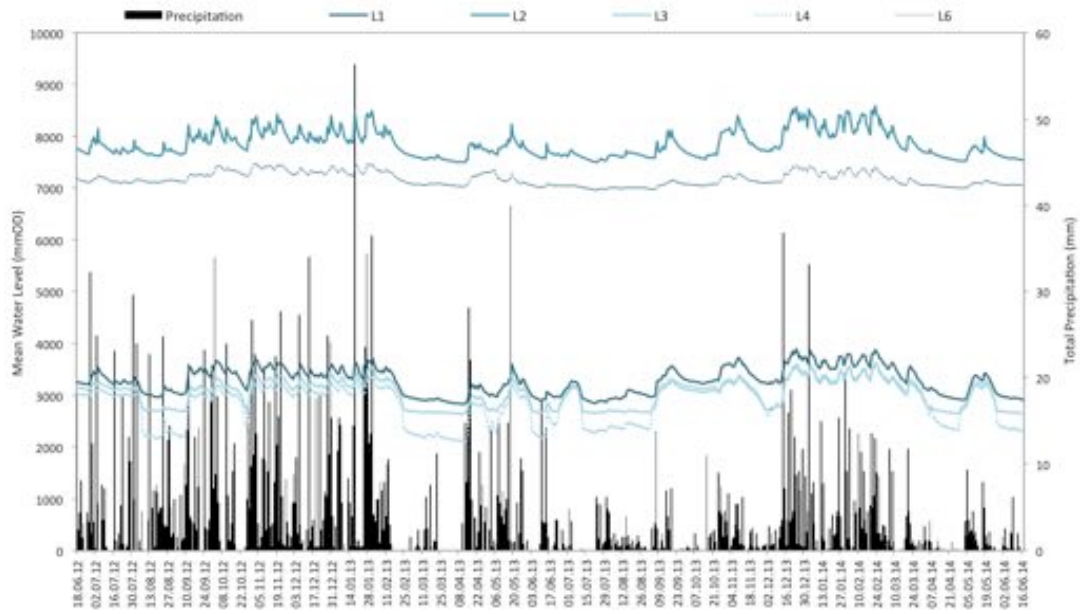
Opened	Closed	Duration Open (days)	Duration Closed (days)	Reason
-	18 th Jun 2012	-	46	NPWS Hydrological Management
3 rd Aug 2012	10 th Sep 2012	38	39	NPWS Hydrological Management
19 th Oct 2012	27 th Oct 2012	8	118	Landowner Adjustment
22 nd Feb 2013	9 th Apr 2013	46	18	Landowner Adjustment
27 th Apr 2013	17 th May 2013	20	13	NPWS Hydrological Management
30 th May 2013	25 th Jun 2013	19	17	NPWS Hydrological Management
10 th Jul 2013	5 th Sep 2013	58	77	NPWS Hydrological Management
20 th Nov 2013	14 th Dec 2013	24	105	NPWS Hydrological Management
28 th Mar 2014	1 st May 2014	34	24	NPWS Hydrological Management
23 rd May 2014	-	26	-	NPWS Hydrological Management

4.4.2.1 Two-year hydrological survey: seasonal trends

Daily average water levels relative to Ordnance Datum (OD), measured at the hydrological monitoring sites over the two-year survey period reveal considerable seasonal variations in water level across the lake, wetland and dune system that are strongly aligned with the recorded fluctuations in precipitation (Figure 4.30). The envelope of water level variability

across lake and river sites (L1-4 and L6) during the two-year survey period is close to a metre (Figure 4.31): the sluice/outflow (L4) has a range of 1251 mm, the Abberachrin river (L2) a range of 1000 mm, the Duvoge river (L1) and lake (L3) ranges of 875mm and 900mm, respectively. Sandfield Lough (L6) displays the smallest water level range throughout the monitoring period, only varying by 454mm. Water levels measured at all of the lake sites exhibit gradual seasonal increases, from June 2012 to late January 2013 and from September 2013 to February 2014, and subsequent steady declines, from February 2013 to August 2013 and from March to June 2014.

a)



b)

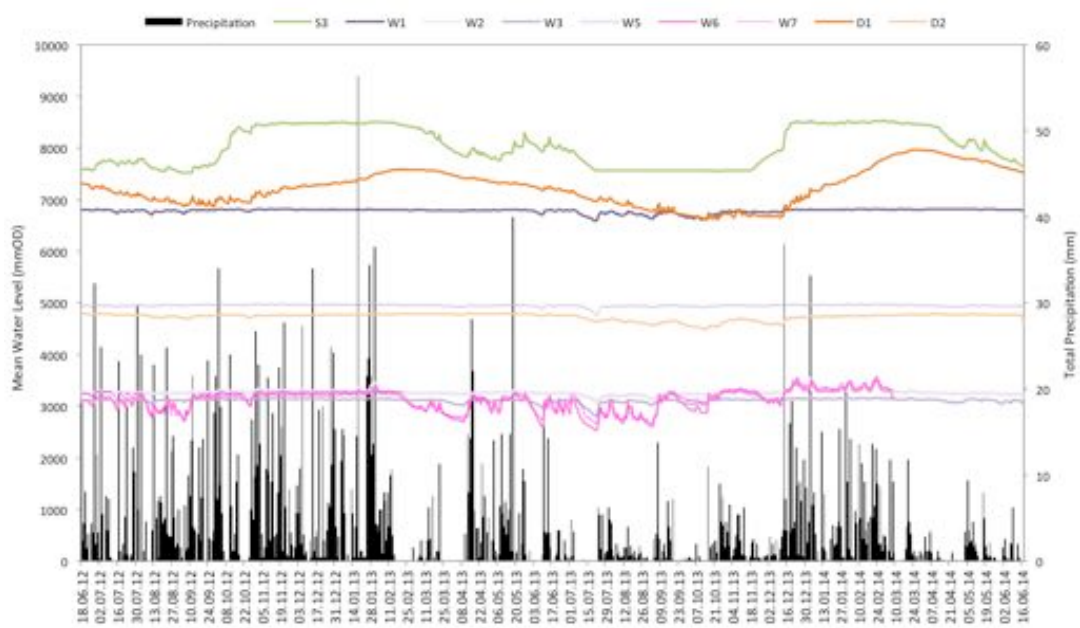


Figure 4.30 Daily total precipitation (mm) measured at the on-site weather station and daily mean water levels (mm) relative to Ordnance Datum (OD) at Malin Head, in the a) lake system (L1-4; L6) and b) in the wetland system (S3; W1-3; W5-7; D1-2) recorded during the two-year monitoring period from 18th June 2012 to 17th June 2014.

Within this overall seasonal trend of gradual rises and falls in water level are frequent (weekly to bi-weekly), sharp fluctuations in daily average water level of up to 300mm. These are in response to regular and often fairly large precipitation events of over 100mm per day, which increase in frequency and intensity throughout the autumn to winter period. Water levels at the wetland and dune sites range by 993mm at S3 (northwest extent of the wetland), 754mm at W6 and 657mm at W7 around the lake margins, and 722mm in the dune blowout (D1). It should be noted that data are absent for the two sites next to the lake (W6 and W7) from 7th March to 17th June 2014 as divers were reallocated. In contrast, the beach (D2) and western wetland sites (W1-3) show very little seasonality, maintaining fairly constant water levels throughout the entire monitoring period with ranges of less than 200mm. This suggests that water levels at these sites are maintained by near-continuous hydrological recharge. D1, located in the dune blowout, and S3, located to the northwest of the wetland, exhibit purely independent seasonal and flashy water level patterns that peak during the winter and spring, respectively.

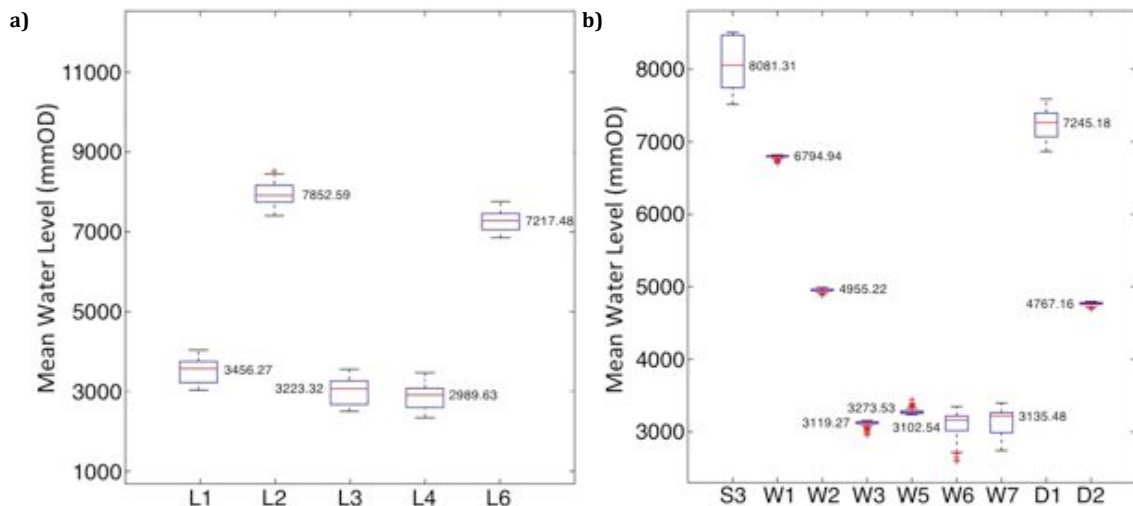


Figure 4.31 Range in mean daily water levels (mm) relative to Ordnance Datum at each site across a) the lake system at L1-4 and L6 and b) the wetland system at S3, W1-3, W5-7 and D1-2, during the two-year monitoring period from 18th June 2012 to 17th June 2014. Note the overall mean water level (mmOD) for the survey period is labelled for each site. Note differences in y-axis scales.

Also superimposed on the overall two-year seasonal water level trend are five large and rapid drops in water level of up to 1m that occur over a period of less than a week. These occur from: August to mid-September 2012; mid to late October 2012; mid-February to early April 2013; during late April 2013; and mid-May to early June 2013 and are most evident at the lake (L3), the outflow/sluice (L4) and the Duvoge river (L1), and are attributed to sluice adjustment. For example, during the August 2012 to mid September period, water levels in Sandfield Lough (L6) remain at a relatively consistent elevation. In contrast, water levels in Sheskinmore Lough (L3) fall dramatically and remain at a lower elevation throughout the period, receiving significant inputs from the inflowing rivers. Water level management was confirmed as the cause, as on the 3rd August 2012 the National Parks and Wildlife Service (NPWS) opened the sluice, closing it again on 10th September 2012 (Emer Magee pers. comm. 8th November 2013).

It is clear that sluice adjustment is the primary cause of the rapid and significant declines in water levels in Sheskinmore Lough (L3), which exhibits a very different hydrological regime to the unsluiced Sandfield Lough (L6) (Figure 4.32). Indeed, dates of opening and closing recorded by the NPWS coincide with the rapid and significant drops in water level and also the steep rises in water level highlighted in Figure 4.32. For example, the hydrological impact of the NPWS opening the sluice on 3rd August 2012 resulted in a significant fall (600mm) in water level in less than 7 days within Sheskinmore Lough. This pattern is repeated at eight more stages throughout the survey period, with each sluice adjustment resulting in similar scale shifts (~600mm) in water level within a timeframe of less than a week.

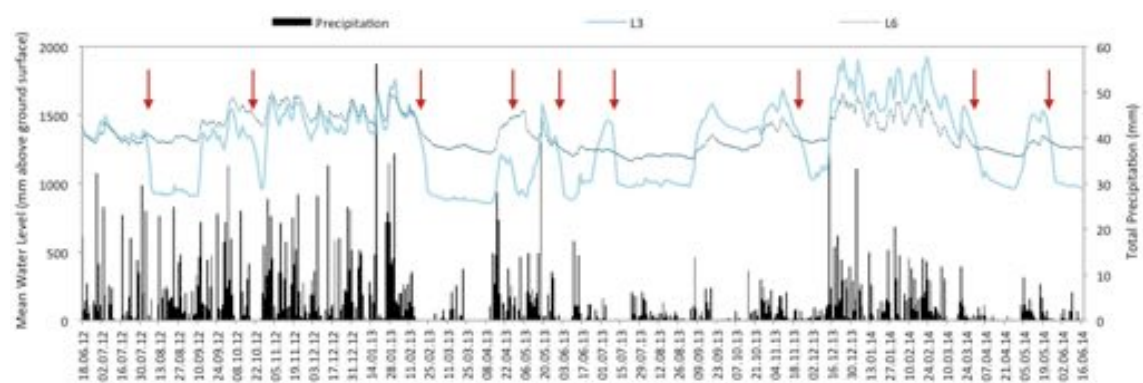


Figure 4.32 Daily total precipitation (mm) measured at the on-site weather station and daily mean water levels relative to Ordnance Datum (OD) at Malin Head (mm) in the a) hydrologically managed Sheskinmore Lough (L3) and b) unmanaged Sandfield Lough (L6) during the two-year monitoring period from 18th June 2012 to 17th June 2014. Note: the red arrows indicate where the sluice was opened during the survey period.

Examination of the mean daily water levels relative to the ground surface at each site presents an alternative context for analysis that emphasises coherence (or lack of) in seasonal variability (Figure 4.33, Figure 4.34). The Abberachrin river inflow (L2) displays the ‘flashiest’ pattern in daily average water levels, exhibiting a typical sensitive riverine response to precipitation events with rapid and short-lived increases and decreases in water level. This pattern is especially prominent throughout the winter months, and is virtually non-existent during spring 2013 due to the near complete absence of precipitation. In contrast, at the Duvoge river inflow (L1) water levels are far less ‘flashy’, displaying more gradual, smaller increases in water level during the same period, a signal that is more similar to Sheskinmore Lough (L3). This is likely a result of the proximity of the monitoring station to the lake itself (Baxter, 1977; Parker et al, 2004). Site L2 (Abberachrin river) is located 1.5km from the lake, whereas site L1 (Duvoge river) is located just 0.75km from the lake, and is possibly experiencing a hydrology driven by lake basin waters backing up into this valley.

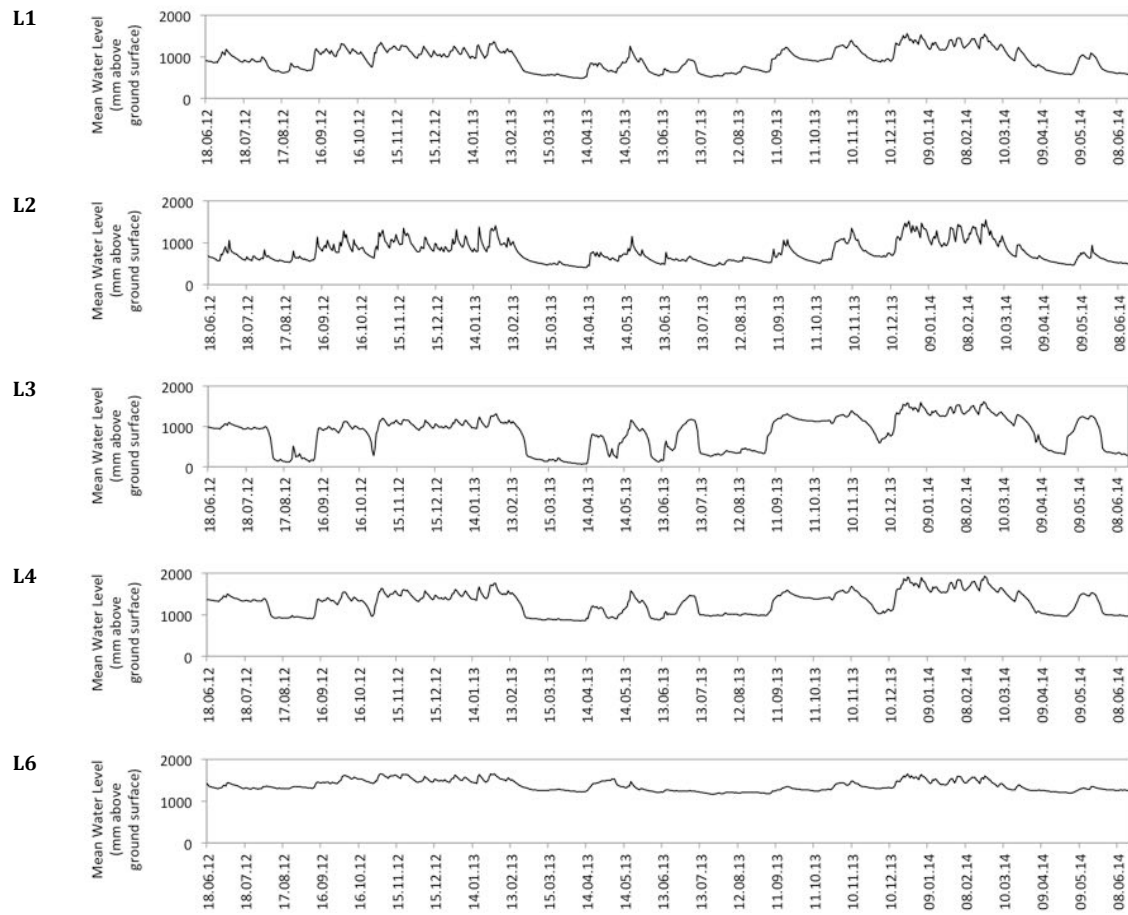


Figure 4.33 Daily mean water levels relative to the ground surface (mm), measured in the lake system at L1-4 and L6 during the two-year monitoring period from 18th June 2012 to 17th June 2014.

Figure 4.34 also emphasises some interesting seasonal water regime patterns across the wetland and dune survey area, especially as it reveals when and where below ground water level breaches the ground surface. At site S3, for example, the water level exceeds the ground surface by up to 200mm from late October 2012 to late February 2013, despite water levels falling to below 400mm depth during the summer to autumn period in 2012 and the spring to autumn period in 2013. In contrast, wetland sites W1, W2 and W3 exhibit water levels that never breach the ground surface beyond a couple of mm depth. These sites do, however, maintain a water level that is fairly consistent throughout the two-year survey period, remaining at or just below the ground surface, except for during summer to early autumn period. In contrast, sites W5, W6 and W7 flooded above their ground surface water level state during the winter months. W6 and W7, situated close to the lake, display very similar water level regimes to the lake (L3) and sluice (L4). D1 in the dune blowout and S3 to the northwest of the wetland, on the other hand, exhibit seasonal patterns in water level regime, rising above (>100mm) the ground surface in the winter and spring and falling well below (>400mm) the ground surface in the summer and autumn months. D2, situated in the beach pond, remains flooded throughout the two-year period at a level of approximately 300mm depth, except during the autumn months when water depths decline, especially during the autumn of 2013.

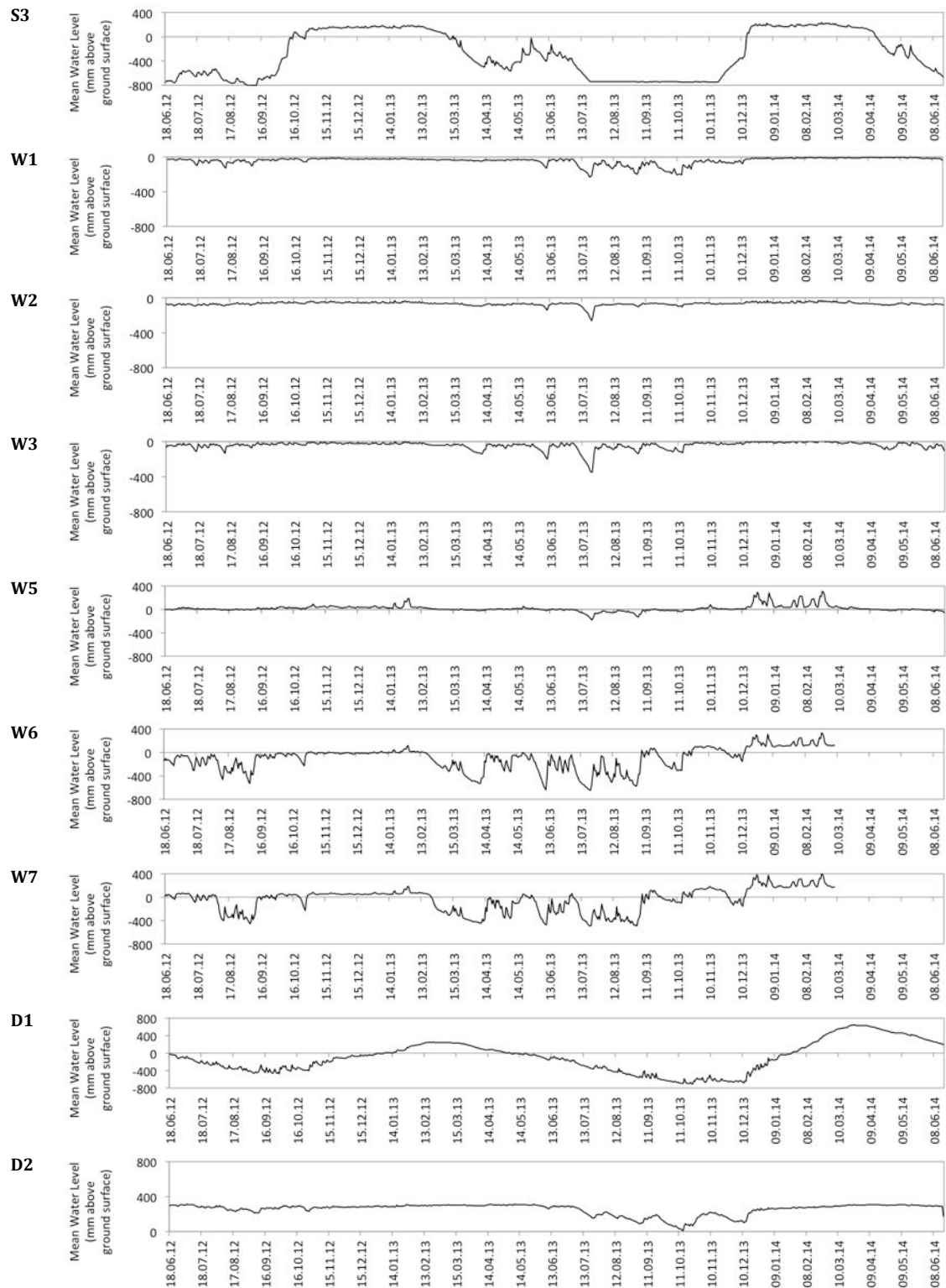


Figure 4.34 Daily mean water levels relative to the ground surface, measured during the two-year monitoring period from 18th June 2012 to 17th June 2014 in the dipwells installed across the wetland system (S3; W1-3; W5-7; D1), and in the hydrological monitoring site in the dune pond (D2). Note differences in y-axis scales.

4.4.2.2 Six-week hydrological survey: the role of precipitation

Hourly average water levels relative to Ordnance Datum (OD) were measured over a discrete six-week period from 7th September to 22nd October 2012, in order to more accurately gauge the role of precipitation on the site hydrology (Figure 4.35). The shorter monitoring timeframe also reveals more details about the hydrology of peripheral sites, and their interaction with the core lake and wetland system. L2 within the Abberachrin river displays the most variable hydrological regime during this period with a range of 709.93mmOD, responding most sensitively to precipitation, especially during high rainfall events on 12th September, 2nd October and 11th October (Figure 4.36). In contrast, L1 within the Duvoge river responds to the same precipitation events with a comparatively more suppressed intensity and a range of 644.52mmOD that is more akin to the water level range within the lake at L3 (632.47mmOD). The water level regime of L4 at the sluice displays the greatest range in water level varying between 968.04mmOD during the six-week monitoring period. At the other extreme, L6 in Sandfield Lough reveals a more constant hydrology, only varying by 318.63mmOD in water level.

Water levels at the western end of the wetland remain relatively constant during the six-week monitoring period, with W1, W2, W3, W4 and W5 only varying between 31.27mmOD and 51.43mmOD, despite a number of large precipitation events. In contrast, at the eastern end of the wetland, sites by the lake (W6 and W7) rise from a water level low of around 2840mmOD on the 8th and 9th October 2012 to a high of around 3250mmOD on 5th October 2012, with a range of 440mmOD. The hydrology also varies across the dune system, with water levels ranging by 200.63mmOD in the dune blowout (D1) and by just 83.58mmOD in the beach pond (D2). In the areas surrounding the lake and wetland system, water level regimes vary considerably. To the northwest of the wetland, water levels at S3 vary as much as 881.43mmOD but at S2 they vary only by 248.67mmOD. To the southwest end of the wetland the water level varies by just 56.30mmOD.

In accordance with the two-year daily data, the six-week hourly data also reveal similarities in water level between the Duvoge river at L1 and the lake (L3). The hourly data, however, also reveal two noticeable differences. Firstly, L1 retains a riverine hydrological signal unapparent at L3. L1 is more responsive to low intensity rainfall events, partly as this section of the Duvoge is on a gradient that facilitates downstream flow. The lake (L3), in contrast, is situated over basin area topography where downstream flow is minimal. Reduced channel area upstream of the lake also means the same amount of precipitation results in greater volumes of water at L1 as accommodation space is more limited (Euliss & Mushet, 1996; Coops et al, 2003). Secondly, water levels in the Duvoge respond more readily than the lake to high intensity and low intensity-high frequency precipitation events. For example, the high intensity (>20mm) rainfall event on 30th September resulted in a 100mm increase in water level in the Duvoge in only 12 hours, compared to a 100mm increase in over 18 hours in the lake. Again, this response is a function of limited accommodation area; however in the case of larger, prolonged precipitation events, water level is determined by catchment area and substrate saturation, as well as by channel cross section (Gibson et al, 2006; Coops & Hosper, 2009).

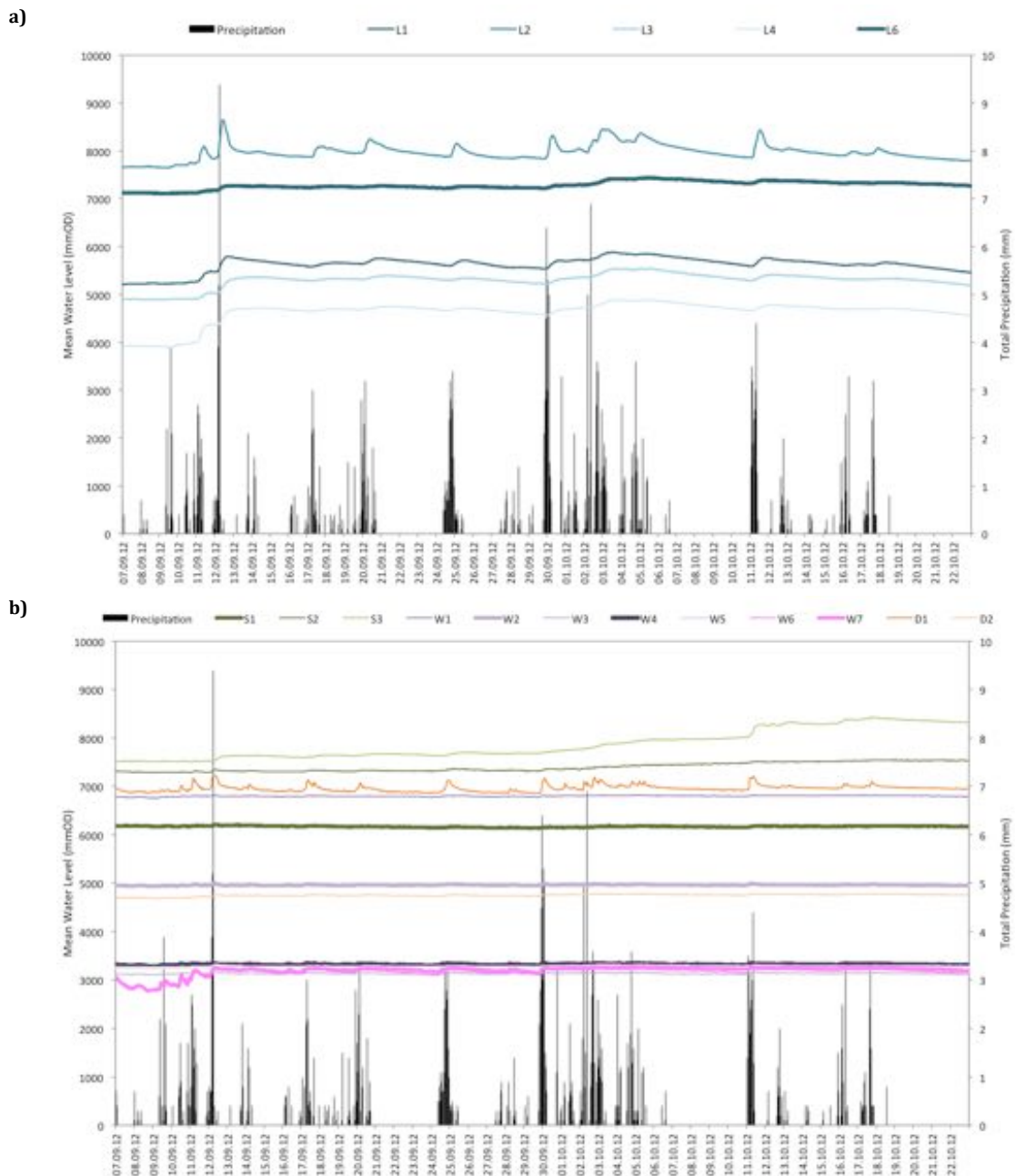


Figure 4.35 Hourly total precipitation (mm) measured at the on-site weather station and hourly mean water levels (mm) relative to Ordnance Datum (OD) at Malin Head, in the a) lake system (L1-4; L6) and b) in the wetland system (S1-3; W1-7; D1-2) recorded during the six-week monitoring period from 7th September 2012 to 22nd October 2012.

The installation of an AQUAlogger 520PT10 on the southern side of Sheskinmore Lough during the six-week monitoring period, to develop a more detailed understanding of the hydrology of the lake and wetland system, further supports these findings (See Appendix C. for full report). The AQUAlogger also revealed the water levels in the Duvoige are very similar to those recorded in the lake. The correlation between the Duvoige water level at L1 and the lake water level at L3 is strong, displaying a positive correlation ($r^2 = 0.92$; $p = 0.009$). In addition, the AQUAlogger data indicated the more precipitation-sensitive riverine hydrology at L1 is dampened by limnological hydrology due to the close proximity of the site to the lake and the wider peaty valley. L2 in the Abberachrin river, on the other hand, exhibits a more 'flashy' water level regime typical of riverine response to

precipitation due to its narrow bedrock valley and distance from the lake which precludes any influence from limnological hydrology (O'Sullivan & Reynolds, 2003; Alemayehu et al, 2006). Consequently, water levels in the Abberachrin increase rapidly following precipitation events, especially when rainfall exceeds 10mm for a period of more than an hour. Water levels in this river also decline more rapidly during dry periods than those in the Duvoge, lake and outflow. As a general rule it can be said that the higher the rainfall, the flashier the response of the Abberachrin river, with increased levels of 500-1000mm following intense rainfall events (>15mm) as opposed to 200mm increases following minimal rainfall events (<15mm).

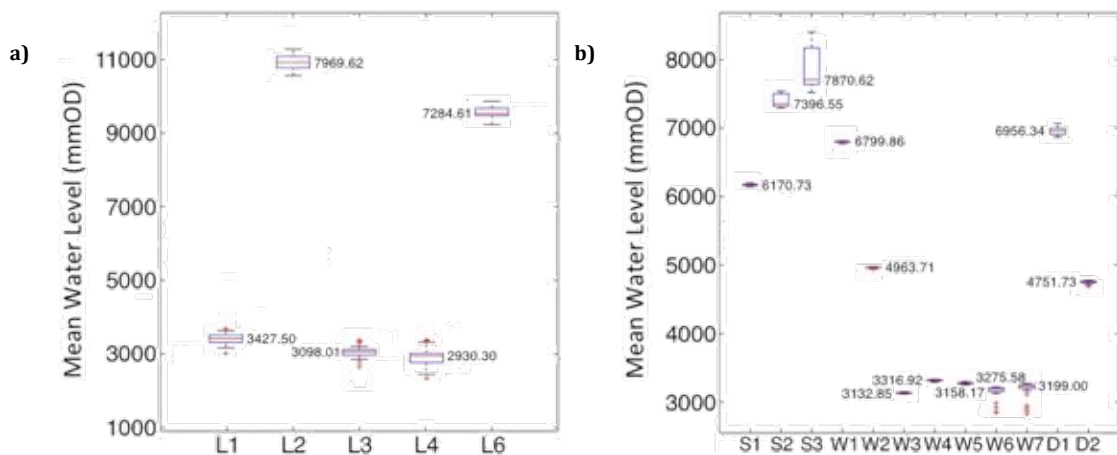


Figure 4.36 Range in hourly mean water levels (mm) relative to Ordnance Datum at each site across a) the lake system at L1-4 and L6 and b) the wetland system at S1-3, W1-7 and D1-2, during the six-week monitoring period from 7th September to 22nd October 2012. Note the overall mean water level (mmOD) for the survey period is labelled for each site and also note the differences in y-axis scales.

Examination of the hourly average water levels relative to the ground surface at each hydrological monitoring site sets the six-week data in a context from which the role of precipitation can be more effectively assessed (Figure 4.37). There are clear similarities and differences in water level responses to precipitation between the sites in and around the lake and those located across the wider wetland system. For example, overland water sites in the inflowing rivers (L1 and L2), the lakes (L3 and L6), the outflow (L4) and those close to the lake (W6 and W7) respond to the large precipitation events on 30th September 2012 and 3rd October 2012 with noticeable increases (>100mm) in water level within 24 hours following the rainfall events. In addition, the effect of the closing of the sluice on 10th September 2012, is unmistakable in L1, L3 and L4. In contrast to the lake sites, groundwater sites across the wider wetland (W1-5) display comparatively small increases in water level (<20mm) in response to these precipitation events. Beyond the wetland, site responses to precipitation vary further, with small (<50mm) increases in above ground water level on 30th September 2012 and 3rd October 2012 in the dune ponds at S1 and S2, and in the beach pond at D2. Although D1 in the dune blowout is a groundwater site for most of the year, it displays flashy water level fluctuations that are more indicative of the overland water level pattern observed in the Abberachrin river at L2. This turbulent water level pattern therefore indicates the dune pond hydrology is extremely sensitive to precipitation events in the sub-daily context.

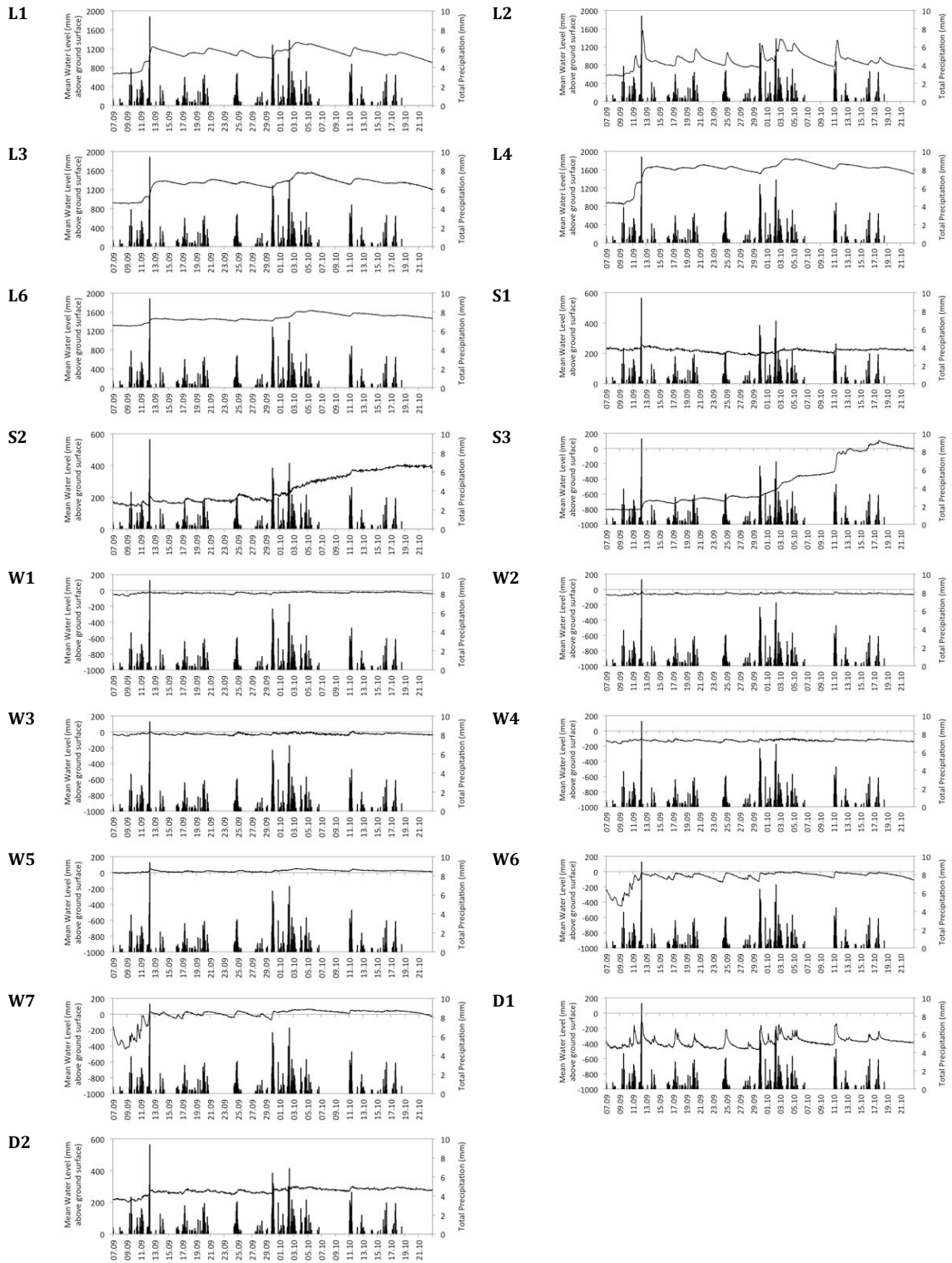


Figure 4.37 Hourly total precipitation (mm) measured at the on-site weather station and hourly mean water levels relative to the ground surface (mm), measured in the lake system at L1-4 and L6, across and around the wetland and dune system at S1-3, W1-7 and D1-2 during the six-week monitoring period from 7th September to 22nd October 2012. Note differences in y-axis scales.

4.4.2.3 Four-week hydrological survey: the role of the sluice

Hourly mean water levels were also measured over a discrete four-week period during the summer from 1st to 31st July 2013, in order to assess the effect of the sluice on the hydrology across the lake, wetland and dune system in more detail. On 10th July 2013, the sluice was opened by the NPWS. Close examination of the water levels across the lake system reveal that the Duvoge river (L1), Sheskinmore Lough (L3) and the outflow river at the sluice (L4) all respond immediately to sluice manipulation, and within 48 hours, water levels at these sites had fallen to a new lower level that is maintained for the remainder of the four-week period (Figure 4.38). This response is also evident in the large water level ranges at L1 (854mm), L3 (983mm) and L4 (1273mm) during the four-week period (Figure 4.39). In contrast, Sandfield Lough (L6), which is not subject to hydrological management, does not display the same fall in water level and exhibits a much narrower range (<80mm). There is a clear difference in the pattern of water level between Sheskinmore Lough and Sandfield Lough with the latter retaining a relatively constant water level throughout the survey period. In contrast, W5 and W6, which are situated close to the lake, do not appear to respond to the sluice opening.

Further away from the lake, the observed water level regimes do not appear to respond to the sluice manipulation, especially when observed relative to the ground surface (Figure 4.40). In the dune ponds surrounding the western end of the wetland (S1 and S2), water levels exhibit a gradual steady decline (160mm and 225mm, respectively) throughout the 31-day survey period and do not appear to be influenced by sluice operation. The dune blowout and dune pond (D1 and D2) also display similar hydrological regimes during this period, with D1 decreasing by 209mm over the 31 days and D2 showing the most gradual and smallest variation in water level of only 140mm, but no noticeable decline following 10th July 2013. At S3, to the northwest of the wetland, similar water level declines are also evident; however at this site the water levels plateau on the 21st July at 800mm and remain constant for the remainder of the survey period. Water level declines within the wetland are more marked, decreasing until the 21st and 22nd July at the western end (W1: 220mm; W2: 208mm; W3: 341mm) and also by the lake (W5: 187mm; W6: 620mm; W7: 564mm). It is only W7 on the south side of the lake, where the water level range exceeds 560mm, that any clear response to the sluice is evident.

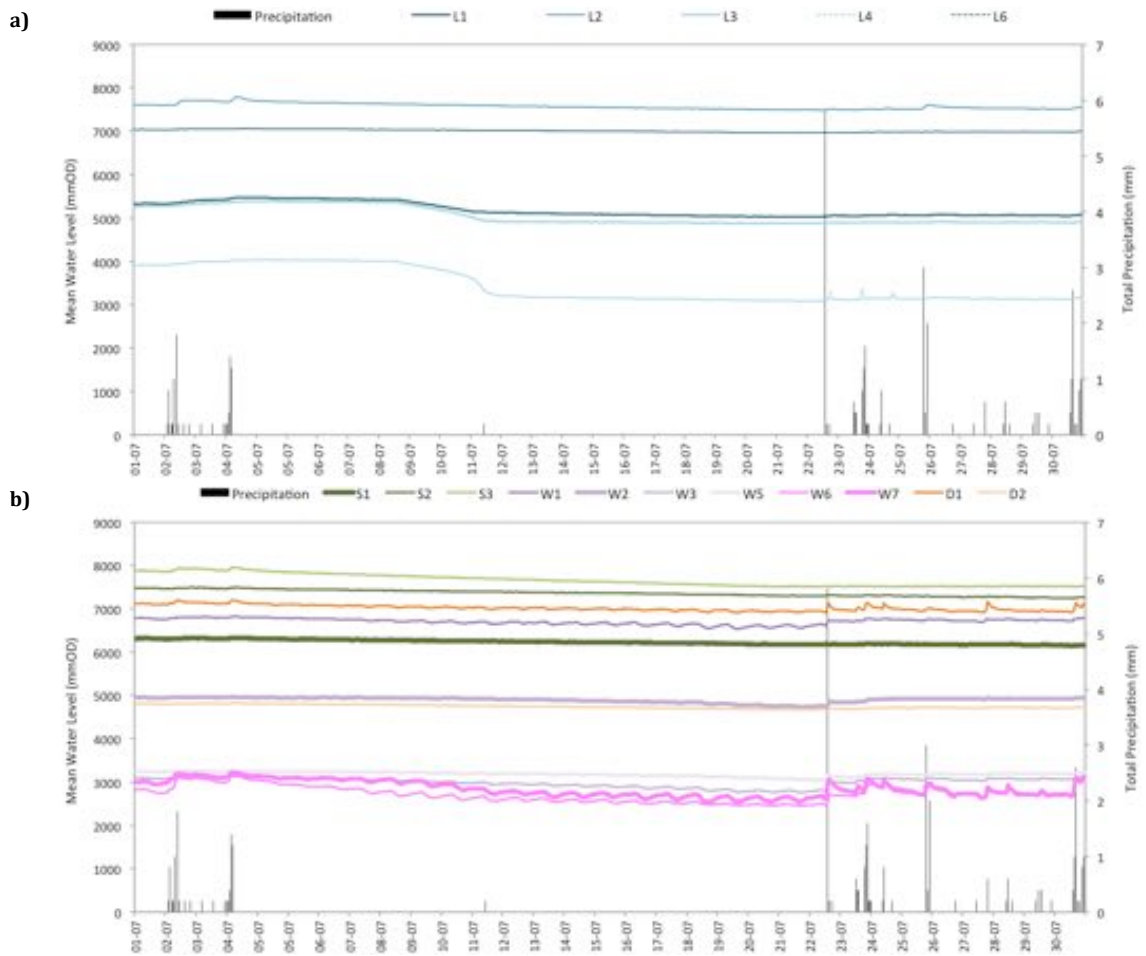


Figure 4.38 Hourly total precipitation (mm) measured at the on-site weather station and hourly mean water levels (mm) relative to Ordnance Datum (OD) at Malin Head, in the a) lake system (L1-4; L6) and b) in the wetland system (S1-3; W1-3; W5-7; D1-2) recorded during the four-week monitoring period from 1st to 31st July 2013.

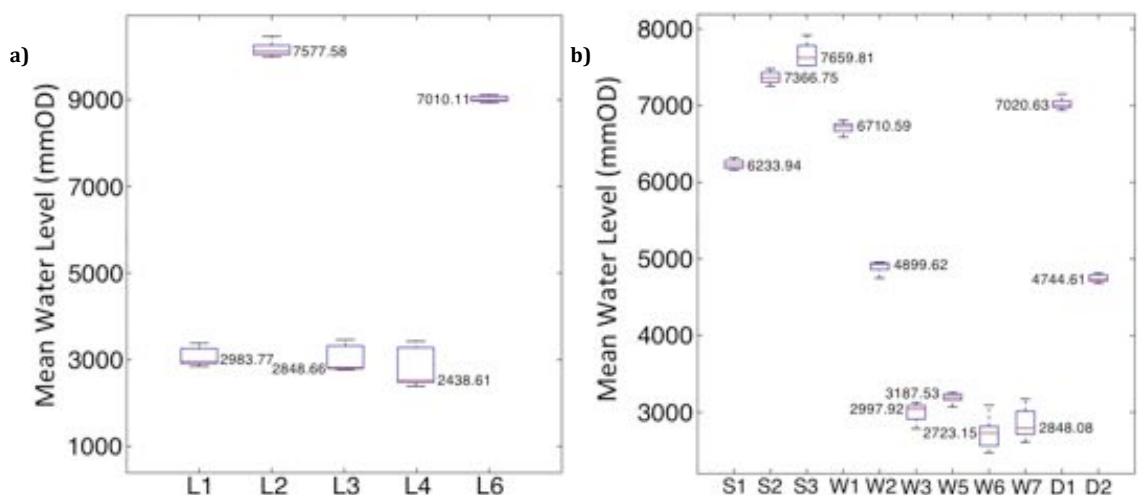


Figure 4.39 Range in hourly mean water levels (mm) relative to Ordnance Datum at each site across a) the lake system at L1-4 and L6 and b) the wetland system at S1-3, W1-3, W5-7 and D1-2, during the four-week monitoring period from 1st July to 31st July 2013. Note the mean daily average water level (mmOD) is labelled for each site and also note the differences in y-axis scales.

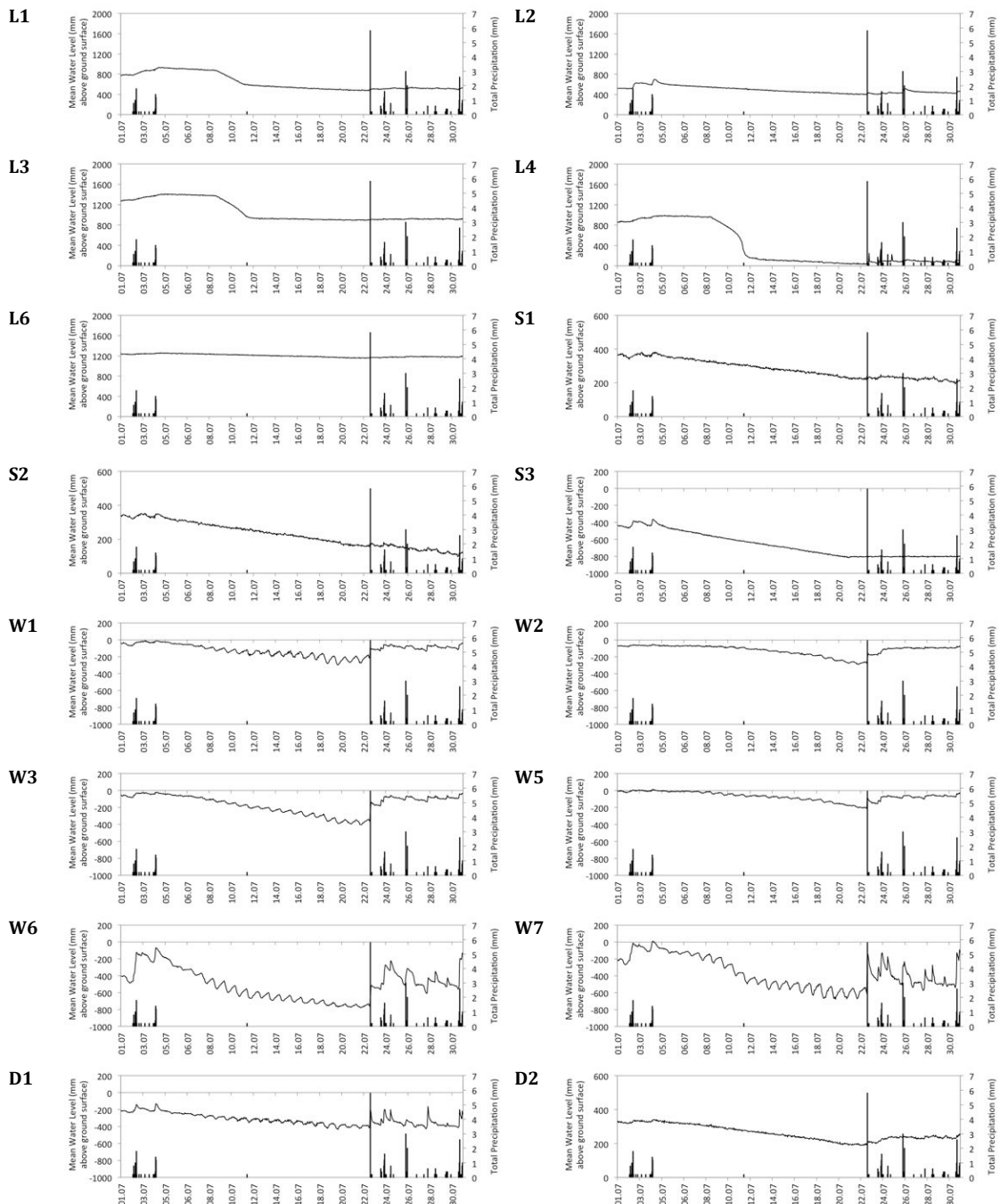


Figure 4.40 Hourly total precipitation (mm) measured at the on-site weather station and hourly mean water levels relative to the ground surface (mm), measured in the lake system at L1-4 and L6 and in the dipwells installed across the wetland system (S1-3; W1-3; W5-7; D1-2) during the four-week monitoring period from 1st to 31st July 2013. Note differences in y-axis scales.

4.4.2.3 Seven-day hydrological survey: spatial connectivity of the system

A seven-day hydrological survey was undertaken from 18th to 24th June 2012 to assess the finite hydrology of the system and, more specifically, to investigate the spatial connectivity of the lake and wetland system. The distribution and elevation of the hydrological monitoring sites used during the seven-day monitoring period are displayed in Figure 4.41. As previously discussed, elevation across the site varies greatly. It is important,

however, to note the hydrological monitoring sites that did not feature in the longer-term surveys (except for W4 which did feature in the six-week survey). For example, there is a 4m difference in elevation between the beach pond (D2) and the wet grassland sites S4 and S5. At elevations of 8.54mOD and 8.19mOD, respectively, S4 and S5 are at the highest elevations, dividing the Sheskinmore Lough basin from the Sandfield Lough basin. The lowest hydrological monitoring site is the outflow site L5 (1.56mOD), located downstream of the sluice. Site W4 emphasises the subtle variations in elevation as, although W3 and W4 are only 100m apart, W4 is 0.28mOD higher than W3 despite the overall easterly decrease in elevation across the wetland.

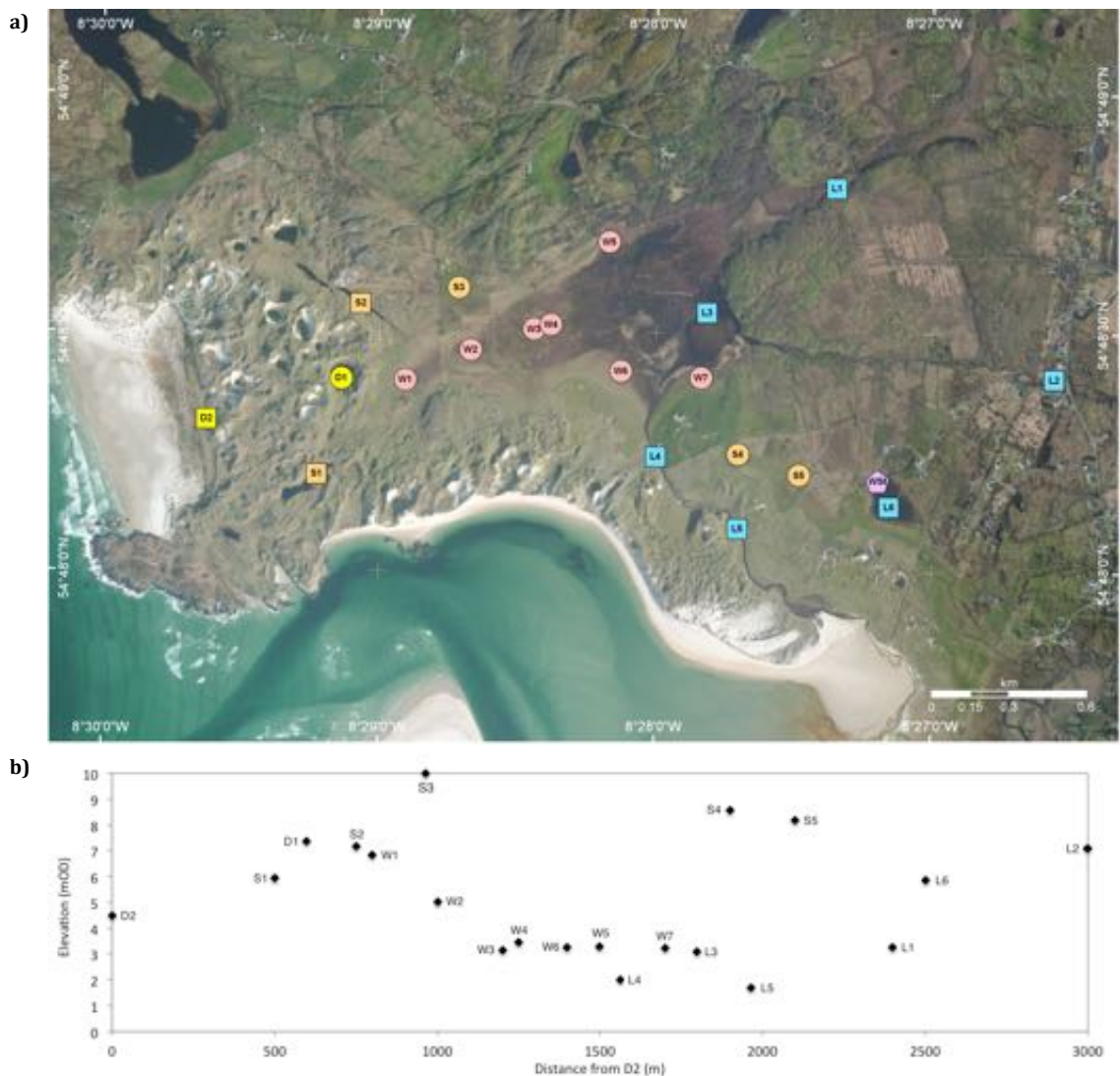


Figure 4.41 a) The location of the on-site weather station (WSt) and the hydrological monitoring sites contributing data to the seven-day survey (18th to 24th June 2012). The sites include those in the inflows (Duvoge L1; Abberachrin L2), Sheskinmore Lough (L3), the sluice outflow (L4), the downstream estuary section of the outflow (L5), Sandfield Lough (L6), the wetland system (W1-7), the dune ponds (S1-2), the surrounding wet grassland (S3-5), the dune blowout (D1) and the beach pond system (D2). b) Elevational distribution of the hydrological monitoring sites from D2 across the lake and wetland system from west to east for the seven-day survey period.

Quarter-hourly average water levels were measured over a discrete seven-day period during the summer from 18th to 24th June 2012 in order to capture more subtle hydrological variations across the lake, wetland and dune system. Figure 4.42 and Figure 4.43 show that water levels across the lake system are fairly constant throughout the seven-day period, with only minor sub-daily fluctuations (<20mm) apparent despite a number of moderate precipitation events. Water levels in the Abberachrin river (L2) and Sandfield Lough (L6) display the greatest ranges in water level (79.88mmOD and 107.90mmOD, respectively), while the water level at the sluice (L4) varies the least (28.94mmOD). Water levels across the wetland and dune system display a similar pattern of constant levels, with only those sites by the lake (W6 and W7) and the between-lake wet grassland sites S4 and S5 varying by more than 60mm. These sites, along with W3, appear to be the most sensitive to the larger precipitation events that occur on 18th June 2012.

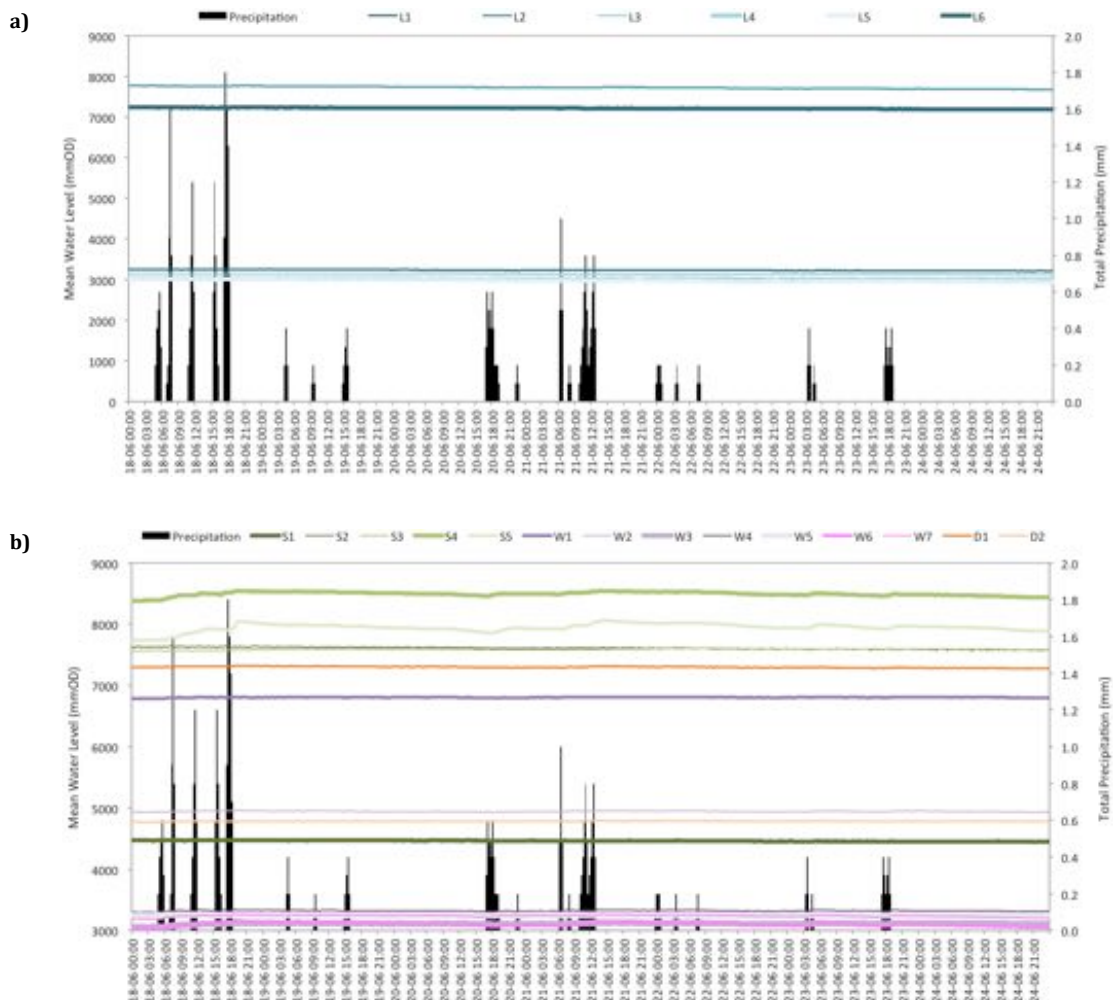


Figure 4.42 Quarter-hourly total precipitation (mm) measured at the on-site weather station and quarter-hourly mean water levels (mm) relative to Ordnance Datum (OD) at Malin Head, in the a) lake system (L1-6) and b) in the wetland system (S1-5; W1-7; D1-2) recorded during the seven-day monitoring period from 18th to 24th June 2012.

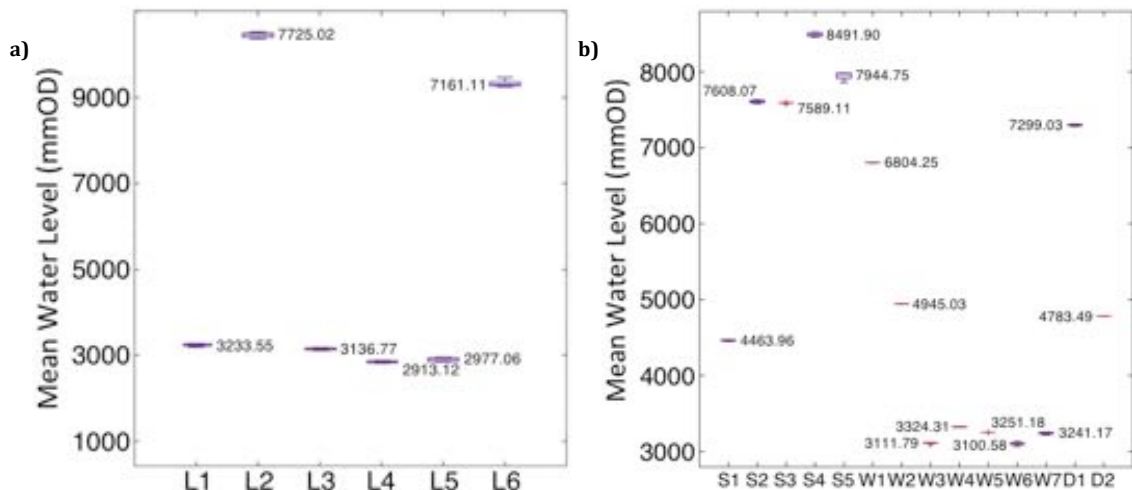


Figure 4.43 Range in quarter-hourly mean water levels (mm) relative to Ordnance Datum at each site across a) the lake system at L1-6 and b) the wetland system at S1-5, W1-7 and D1-2, during the seven-day monitoring period from 18th to 24th June 2012. Note the mean daily average water level (mmOD) is labelled for each site and also note the differences in y-axis scales.

Examination of the water levels relative to the ground surface during the seven-day survey period further emphasises spatial variations in hydrology. Figure 4.44 reveals that the water level in the lake system decreases slightly throughout the survey period. This decrease is interrupted by small fluctuations in water level that occur every 2-3 hours. The estuary outflow (L5) exhibits the same gradual decrease; however two anomalous peaks (25mm) are apparent at 22:00 on 21st June 2012 and at 22:00 on 22nd June 2012. It should be noted that the sluice was closed throughout this survey period. This is supported by the hydrological behaviour of the unsluiced Sandfield Lough (L6) during this survey period, which also displays an overall consistent decline (50mm) in water levels.

In contrast to the lake system, water levels across the wetland, dunes and surrounding areas do not show the same steady water level decline. Exceptions are the water levels in the dune blowout (D1), surrounding dune pond sites S1 and S2, which decline by 31mm and 64mm, respectively, and W6 and W7 located next to the lake, which display the highest ranges in water level in the wetland (52.45mmOD and 43.87mmOD, respectively). Subsurface water levels at S3, on the other hand, increase by 40mm towards the ground surface over the first 4 days of the seven-day period. Between-lake sites S4 and S5 display an altogether different hydrological regime, increasing from depths of 160mm and 450mm below the ground surface, respectively, to depths of 0mm and 150mm, respectively, within the first 24hours of the survey period. Across the wetland, water levels remain relatively constant throughout the seven-day survey period. In similar fashion to S3, S4 and S5, water levels in the wetland at sites W1-4 and W6-7 display an initial rapid increase in water level on 18th June 2012. The beach pond (D2) exhibits relatively constant water levels, only ranging by 18mm during the 7-day period.

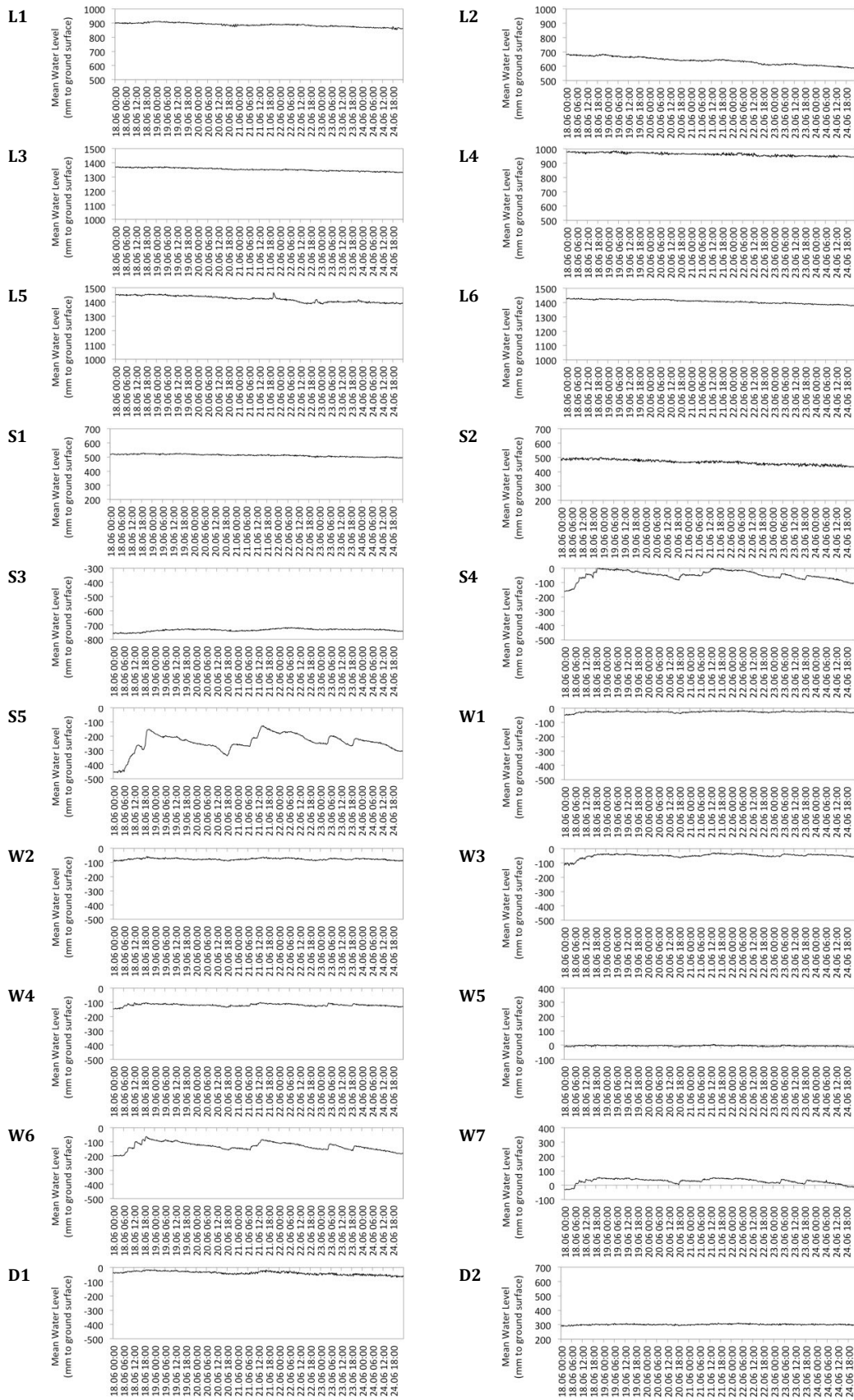


Figure 4.44 Quarter-hourly mean water levels relative to the ground surface, measured during the seven-day monitoring period (18th to 24th June 2012) in the lakes (L3, L6), inflows (L1-2) and outflow (L4-5), dune ponds (S1-2), surrounding wet grassland (S3-5), wetland (W1-7), dune blowout (D1), and beach pond (D2). Note differences in y-axis scales.

It is clear that there are similarities in hydrological behaviour between different hydrological monitoring sites across the lake, wetland and wider system, whilst some parts of the site appear to operate independently. A PCA of the quarter-hourly mean water levels shows that the majority of sites across the wetland, lake and rivers have aligned hydrologies, but that sites D1 (dune pond), D2 (back of beach pond) and S3 (base of Mullyvea) are distinctly different (Figure 4.45). Sheskinmore Lough (L3) and its outflow (L4) display a strong positive relationship ($r^2 = 0.93$; $p = 0.01$). In fact, all of the lake sites (L1-6) are strongly related to each other ($r^2 = >0.80$; $p = <0.05$) and also possess hydrological relationships with the surrounding dune ponds at S1 and S2 ($r^2 = >0.90$; $p = <0.05$). The Duvoge river, for example, is very strongly related to the lake (L3) ($r^2 = 0.98$; $p = 0.009$). In contrast, the Abberachrin river (L2) exhibits strong relationships with Sandfield Lough at L6 ($r^2 = 0.96$; $p = 0.09$).

The relationships between the lake system sites and the wetland, dunes and surrounding sites, however, are more varied. Sites L1-6 exhibit virtually no relationship with the western (W1-2) and central arm (W3-4) of the wetland; however L2 is loosely associated with site W5 to the north side of the lake ($r^2 = 0.74$; $p = 0.03$). L1, L3 and L4 are also related ($r^2 = >0.70$; $p = <0.05$) to W6 and W7 next to the lake. Sandfield Lough (L6), on the other hand, is also strongly related to sites at the western end of the wetland W1-3 ($r^2 = >0.80$; $p = <0.05$) and W5 ($r^2 = 0.88$; $p = 0.08$). Sites within the wetland are related to one another, especially W3 and W6 ($r^2 = 0.99$; $p = 0.001$), W1 and W2 ($r^2 = 0.89$; $p = 0.003$) and W6 and W7 ($r^2 = 0.92$; $p = 0.02$). Site D1 in the dune blowout is the most distinctive site when compared to the other hydrological monitoring sites as it only shows a statistically strong relationship with S3 ($r^2 = 0.93$; $p = 0.05$). In contrast, the beach pond site D2 is strongly related ($r^2 = >0.80$; $p = <0.05$) to the two lakes (L3 and L6), and the wetland sites W1-3 and W5. Although the dune pond D2 displays strong statistical relationships with wetland sites, its most significant relationship is with the dune blowout at D1 ($r^2 = 0.90$; $p = 0.001$) and S3 at the base of Mullyvea ($r^2 = 0.87$; $p = 0.02$).

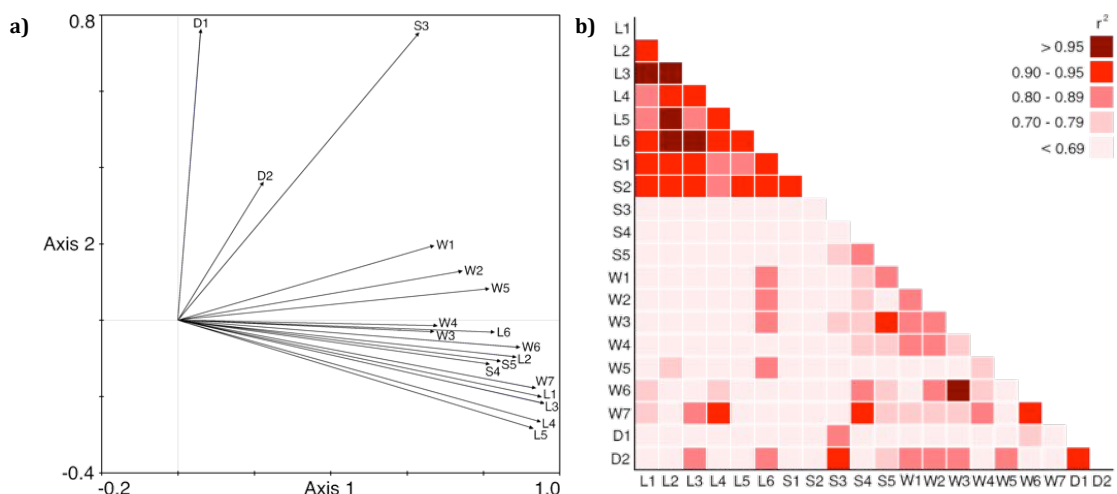


Figure 4.45 a) PCA showing the relationships between the quarter-hourly water levels, and b) linear regression (r^2) values showing the significance of the quarter-hourly water level relationships at the hydrological monitoring sites across the lake (L1-6), wetland (W1-7), dune system (D1-2) and surroundings (S1-5) observed during the seven-day monitoring period from 18th to 24th June 2012. PCA eigenvalues: Axis 1 = 64.3%; Axis 2 = 14.8%.

4.4.3 Hydrological classification of the lake and wetland system

Hydrological classification of lakes and wetlands is a tool increasingly used by conservation managers. By grouping lakes and wetlands based on their hydrology and also their topography and morphology (e.g. average depth, surface area, shoreline configuration), we are better able to manage these habitats in a manner that maximises their conservation potential. Recognizing the distinctions between different lake and wetland system types allows management bodies regionally, nationally and internationally to implement the most appropriate freshwater conservation strategies.

A variety of wetland hydrological classification systems have been recognised, based on broad environmental features such as geological setting, nutrient status and water source (Wheeler & Proctor, 2000). One of the most widespread approaches identifies wetlands based on their hydrotopographical features, specifically their hydrology in relation to situation and shape (Wheeler et al, 2009). In the British Isles, one of the most influential hydrotopographical classifications was that proposed by Goode (1972). It has shaped the approaches of scientists and conservationists such as Ratcliffe (1977), Wheeler (1984) and Lloyd et al (1993), the latter of which defined the hydromorphological classification system based on the main external sources of water and mechanisms of delivery. Lloyd et al (1993)'s scheme comprises seven classes, with groundwater-fed wetlands divided between those supported by confined and unconfined aquifers and classes for wetlands fed by both surface and groundwater (Wheeler et al, 2004).

This study has applied Lloyd et al (1993)'s hydromorphological system to classify the hydrology of the wetland system at Sheskinmore Lough using a combination of hydrological and topographical survey data and field observations (Table 4.5). Topography, or morphology, determines drainage and area within wetlands and therefore also controls the form of wetland development. The hydromorphological system defined by Lloyd et al (1993) indicates the Sheskinmore Lough wetland is an 'Open Water Transition Fen'. Transition fens develop in areas where the water table is at, or exceeds, the ground surface. They are the intermediate 'transitions' between open water where the water table is well above ground for much of the year, and terrestrial habitats where the water table is much lower; as a result they can vary structurally (Kimberley & Coxon, 2013). This variance is evident at Sheskinmore Lough, as they are present in contrasting forms between the open water habitat of the lake and those areas surrounding the northern, western and southern peripheries. For example, to the north and south side of the lake where the immediate topographical gradient is steeper, the wetland is confined to the lake periphery, resulting in a narrow hydrological transition and a fairly even compositional morphology. To the western side of the lake, the flatter morphology of the catchment promotes a wider transition wetland. As the water table is closer to the surface for most of the year in this area, it exhibits varying morphologies including open boggy areas and hummocks separated by wet hollows (Wheeler et al, 2009).

Wheeler & Shaw (1995) recognised that, on their own, wetland hydrotopographical units were useful as broad descriptive units; however for more detailed classification, two

independent layers should be recognised as separate units: situation types and hydrotopographical elements (Table 4.6). The situation type represents the broad and variable landscape situation in which a wetland occurs, while hydrotopographical elements are units with distinctive water supply mechanisms and, often in relation to this, distinctive topographies. Many wetlands, like Sheskinmore Lough, contain a number of hydrotopographical elements and within a particular situation type. In this context, the wetland at Sheskinmore Lough can be defined under two situation types: 'Lakeside Wetland' and 'Coastal Wetland'. Within the 'Lakeside Wetland' unit, the Sheskinmore Lough wetland can be classified as a 'Waterfringe Wetland' element, and it also possesses 'Spring-fed Wetland' characteristics from surrounding hydrological influences.

Classifying the wetland at Sheskinmore Lough purely on the basis of hydrotopography is only useful, however, for separating the habitat into broad descriptive units. For greater detail, this study has also classified the wetland using the hydrogeological classification system defined by Wheeler et al (2004), which identifies the different forms of water transfer that can occur between wetland sites and their surroundings (Table 4.7). Wheeler (2004) made a clear distinction between situation type and water transfer mechanisms for wetland classification, recognising that different transfer mechanisms often occur in different parts of individual sites, thereby pointing away from the development of simplistic whole-site typologies. In this context, the wetland at Sheskinmore Lough hydrology can be classified as 'Class C', due to its surface- and groundwater-dominated hydrology and sandy coastal geology resulting in an unconfined aquifer providing lateral flow. The shallow valley topography and mixed sediment geology consisting of sand, peat and gravel of this wetland type has been cross-correlated with the open water transition fen identified by hydrotopographical classification.

The hydrotopographical and hydrogeological wetland classification systems defined by Lloyd et al (1993) and Wheeler & Shaw (1995) stressed the need to understand how a wetland works. These ideas have been built upon by Wheeler & Shaw (2000), who developed a conceptual '**W**etland **W**ater **S**upply **M**echanism' classification system called WETMECS. The WETMECS system classifies wetlands based on both their hydromorphological and hydrogeological features, essentially conceptual units that describe the supply and distribution of water in wetlands. Specifically, this classification combines landscape situation type, hydrotopographical element, water supply mechanism, base status (pH) and fertility (Wheeler et al, 2009). By classifying wetland hydrology using the WETMEC approach, this study will not only be able to develop a conceptual understanding of the links between the hydrological characteristics of the system and the site ecology and enhance our understanding of the site ecohydrology, it will also help with the assessment and prediction of likely environmental impacts, and therefore inform conservation management of the wetland system.

Table 4.5 Hydromorphological classification of fen wetlands as defined by Goode, 1977 (Adapted from Lloyd et al, 1993). Note the red box indicates the hydromorphological classification of the wetland system at Sheskinmore Lough.

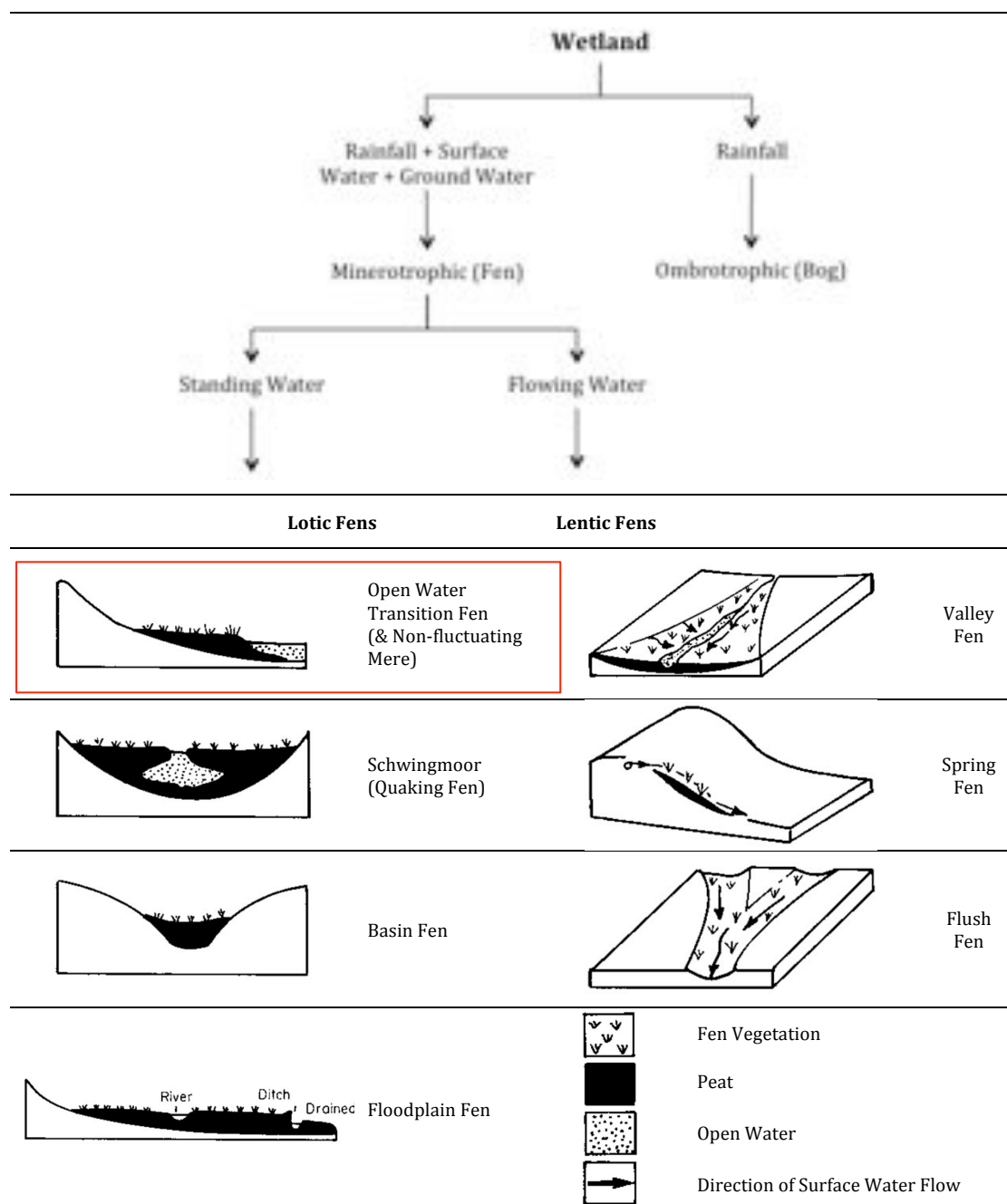


Table 4.6 Wetland situation types and component hydrotopographical elements for wetland classification (adapted from Wheeler & Shaw, 1995). +++ particularly characteristic of the situation type; ++ sometimes occurs within the situation type; + of minor importance, or peripheral.

Hydrotopographical Element	Situation Type					
	Basin Wetland	Lakeside Wetland	Coastal/Floodplain Wetland	Plateau-plain Wetland	Valleyhead Wetland	Hillslope Wetland
Alluvial Wetland			+++		+	
Waterfringe Wetland	+++	+++	++			
Sump Wetland	+++	+++	+++	+++	+	
Percolating Wetland	+++	+	+++	+	+++	
Water Track Wetland	+		++	+	++	
Spring-fed Wetland	++	++	+	++	+++	+++
Run-off Wetland	+	+	+	+	+++	+++
Soakaway					++	+++
Topogenous Bog	+++	++	+++	+++	+	
Hill Bog	+	+	+	+	+	+++

WETMEC associations at Sheskinmore Lough were identified from the WETMEC class descriptions outlined in Wheeler et al (2009) and also in Kimberley & Coxon (2013), which relates specifically to Irish wetlands, using the analysed hydrological data and field observations recorded during this study. These data and observations suggest that the wetland and dune systems at Sheskinmore Lough are best conceptualised by the WETMEC groups listed in Table 4.8 and described in Table 4.9. Figure 4.46 shows the approximate distribution of these WETMEC groups and subgroups across the wetland and dune systems in relation to the hydrological monitoring sites. In the main body of the wetland (W1-4), WETMEC 3, 'buoyant weakly minerotrophic surfaces' or 'transition bogs', dominate. These habitats occur in wet areas with constantly high water tables that are both groundwater-fed and surface-water-fed and are frequently associated with open waters, in this case Sheskinmore Lough itself (EC, 2007). As observed in the field, these habitat areas are varied structurally, from open boggy areas to areas dominated by hummocks interspersed with wet hollows. WETMEC 3 is also associated with the narrow wetland areas close to the lake edges at the southwestern (W6) and southern (W7) sides of the lake where the hydrological transition is narrow and the compositional morphology is more even.

The far western end of the wetland can also be characterised by WETMEC 9 'groundwater-fed bottoms' and WETMEC 12 'fluctuating seepage basins' due to the apparent groundwater influence and water levels that fluctuate episodically close to and above the ground surface. 'Fluctuating seepage basins', WETMEC 12, recognised by Kimberley & Coxon (2013) in Ireland as being associated with machair and dune slack habitats, are associated with those areas located towards the edges of the wetland (S1-5, D-2, and also W1 and W7), where sand substrates are more likely to dominate, promoting the seasonally dynamic water levels observed in the hydrological data. WETMEC 6f, 'Surface Water Percolation Water Fringe' can also be associated with the Sheskinmore Lough wetland system in those areas of the wetland that encroach directly upon open water body of the lake, specifically at sites W4-7. In these areas the ground surface is wet in summer and flooded in winter and the water table usually slightly subsurface and

groundwater is generally unimportant, with surface water making a far greater contribution. Finally, WETMEC 20b 'Percolation Water Fringe' is associated with W5 to the northern side of the lake, where upslope sources are important and the near surface water table is fed mainly by surface runoff.

Table 4.7 Hydrogeological classification of fen wetlands as defined by Lloyd et al, 1993 (Adapted from Wheeler et al, 2004). Note the red boxes indicate the hydrogeological and hydromorphological classification of the wetland at Sheskinmore Lough.

Water Source				
<pre> graph TD WS[Water Source] --> SW[Surface Water] WS --> SWG[Surface Water & Ground Water] WS --> GW[Ground Water] SW --> LF[Lateral Flow] LF --> A[A] SWG --> LA1[Leaky Aquifer] SWG --> UA1[Unconfined Aquifer] LA1 --> LVF1[Lateral & Vertical Flow] LVF1 --> B[B] UA1 --> LF1[Lateral Flow] LF1 --> C[C] GW --> UL[Unconfined & Leaky] GW --> LA2[Leaky Aquifer] GW --> UA2[Unconfined Aquifer] UL --> LVF2[Lateral & Vertical Flow] LVF2 --> D[D] UL --> VF1[Vertical Flow] VF1 --> E[E] LA2 --> VF2[Vertical Flow] VF2 --> F[F] UA2 --> LF2[Lateral Flow] LF2 --> G[G] </pre>				
Class	Input	Topography	Catchment Geology	Cross-Correlation with Hydromorphological Classification
A	Surface Water Runoff Only (with Overbank Flooding)	Often in Topographic Hollow, also Valley or Low Relief Adjacent to River	Clay Predominates	Open Water Transition Fen Schwingmoor Basin Fen Floodplain Fen
B	Leaky Aquifer & Some Surface Water	Shallow Valley	Low Permeability but Mixed (Sand May Exist)	Valley Fen Spring Fen Fluctuating Mire
C	Surface Water & Ground Water from Superficial Deposits	Shallow Valley	Mixed Typical Clay-Sand-Gravel Drift	Open Water Transition Fen Basin Fen Floodplain Fen Valley Fen Spring Fen
D	Groundwater from Superficial Deposits & Underlying Main Aquifer	Valley or Closed Depression	Sands & Gravel Over Clays Over Major Aquifer	Valley Fen Fluctuating Mire
E	Leaky Aquifer	Closed Depression	Clay/Peat Overlying Major Aquifer, Lateral Isolated	Fluctuating Mire
F	Unconfined Main Aquifer	Wide Range	No Superficials, Main Aquifer Rock Outcropping	Valley Fen Spring Fen
G	Unconfined Superficial Aquifer	Shallow Valley	Superficial Sands & Gravels Overlying Clays/Peats	Valley Fen Spring Fen

Table 4.8 WETMEC hydrological classification groups and subgroups associated with each hydrological monitoring site across the central and surrounding wetland (W1-7 and S1-5) and dune (D1-2) systems (Wheeler et al, 2009; Kimberley & Coxon, 2013).

Site	WETMEC Group (Subgroup) ID	WETMEC Group (Subgroup) Name
S1	12	Fluctuating Seepage Basins
S2	12	Fluctuating Seepage Basins
S3	12	Fluctuating Seepage Basins
S4	12	Fluctuating Seepage Basins
S5	12	Fluctuating Seepage Basins
W1	3	Buoyant Weakly Minerotrophic Surfaces ('Transition Bogs')
	9	Groundwater-Fed Bottoms
W2	12	Fluctuating Seepage Basins
	3	Buoyant Weakly Minerotrophic Surfaces ('Transition Bogs')
W3	3	Buoyant Weakly Minerotrophic Surfaces ('Transition Bogs')
W4	3	Buoyant Weakly Minerotrophic Surfaces ('Transition Bogs')
	6(f)	Surface Water Percolation Water Fringe
W5	3	Buoyant Weakly Minerotrophic Surfaces ('Transition Bogs')
	6(f)	Alluvial Floodplain
	20(b)	Percolation Water Fringe
W6	3	Buoyant Weakly Minerotrophic Surfaces ('Transition Bogs')
	6(f)	Surface Water Percolation Water Fringe
W7	3	Buoyant Weakly Minerotrophic Surfaces ('Transition Bogs')
	6(f)	Surface Water Percolation Water Fringe
D1	12	Fluctuating Seepage Basins
	12	Fluctuating Seepage Basins

Table 4.9 Descriptions of the WETMEC hydrological classification groups and subgroups associated with the hydrological monitoring sites across the central and surrounding wetland (W1-7 and S1-5) and dune (D1-2) systems. See Appendix D. for full descriptions (Wheeler et al, 2009; Kimberley & Coxon, 2013).

WETMEC ID & Name	Description
3: Buoyant Weakly Minerotrophic Surfaces ('Transition Bogs')	Quaking, summer-wet surface elevated slightly above telluric water tables; often in basins, over high or low permeability deposits. Near or at surface. May receive weakly telluric water, but precipitation probably a significant component of surface water budget. Groundwater connectivity with aquifers often uncertain. Outflow likely. In some cases may recharge aquifer. Groundwater level often just sub-surface.
6f: Surface Water Percolation Water Fringe	Surface usually wet in summer and flooded in winter. Peat top-layer often loose, and encroaching directly upon open water body. Water table usually slightly subsurface. Fed mainly by surface water, often from dykes connected to watercourses. Groundwater generally unimportant. Directly adjoins water bodies or connected dykes and may contribute to dyke levels, mainly during winter.
9: Groundwater-Fed Bottoms	Troughs or basins, usually on quite deep peat; if on floodplains, isolated from river. Marginal springs or seepages often less evident. Apparently groundwater-fed, but ground water table often well below surface, sometimes because of drainage. Aquifer may be episodically at, above or near surface, but is often low (and more or less in equilibrium with wetland water table).
12: Fluctuating Seepage Basins	Small sumps with strongly fluctuating water table, often from well below surface to flooded, which may relate to aquifer levels. Mainly groundwater-fed. Water table variable, depending on topography and aquifer level; fluctuates strongly, standing water due to topography. Aquifer episodically at, above or near surface. Water level sometimes in (slow) equilibrium with aquifer level, but relationship sometimes obscure.
20b: Percolation Water Fringe	Adjoining open water and receiving water from this, may have different provenance to upslope sources, groundwater importance not clear. Water table at or near surface, fed mainly by surface runoff, some of which sourced by groundwater outflow. Water body irrigates stand. More or less confined or very minor aquifer, or none; sometimes springs and seepages visible, usually well upslope.

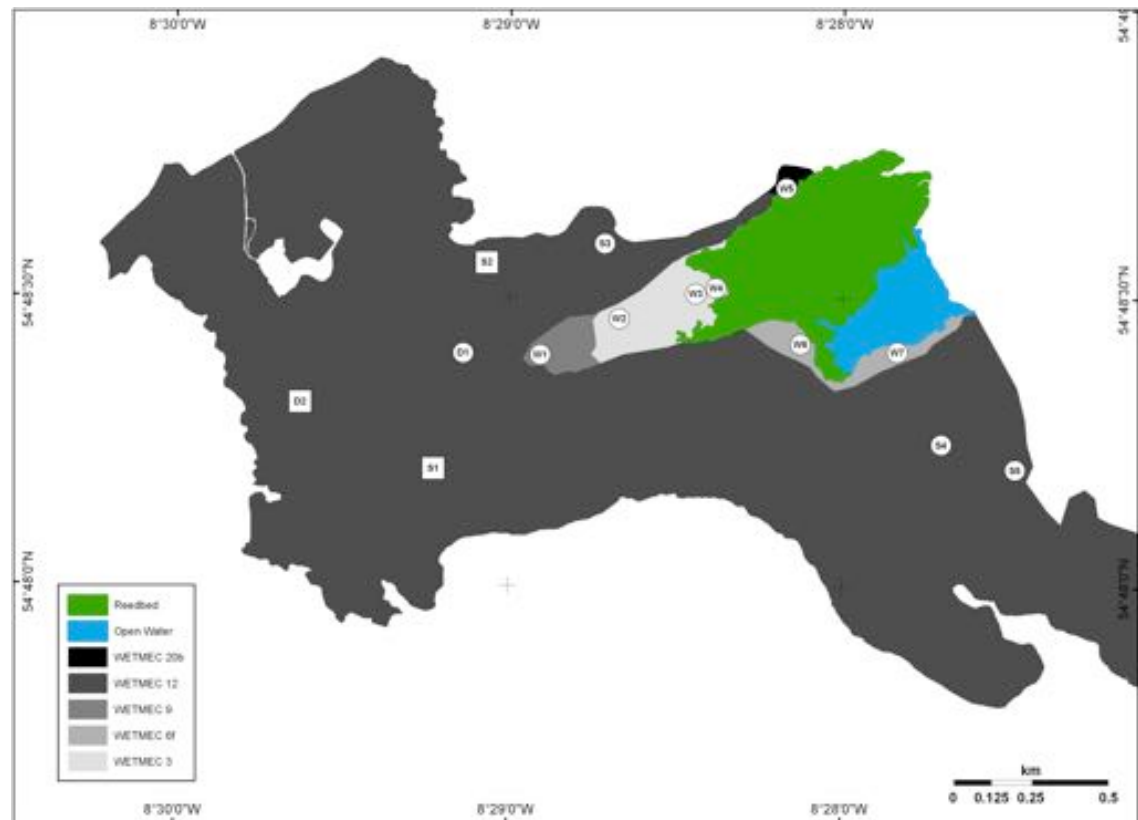


Figure 4.46 Distribution of the WETMEC hydrological groups and subgroups across the wetland and dune systems in relation to the wetland, dune and surrounding hydrological monitoring sites. WETMEC 3: Buoyant Weakly Minerotrophic Surfaces ('Transition Bogs'); WETMEC 6f: Surface Water Percolation Water Fringe; WETMEC 9: Groundwater-Fed Bottoms; WETMEC 12: Fluctuating Seepage Basins; WETMEC 20b: Percolation Water Fringe.

Lakes have been classified on the basis of morphological, chemical and biological variables for many decades (Winter, 1977; Black, 1996; Schiffer, 1998); however classification based on hydrology has only recently been comprehensively developed (UKTAG, 2003; IWFD, 2005; Rowan, 2008a; Rowan, 2008b; UKTAG, 2008; NIEA, 2009; Cui et al, 2010; Rowan et al, 2012). Classifications based on hydrologic characteristics are beneficial as they provide insights into the physical, biological and chemical cycles, as well as responses to impacts and management. Prior to the study of Winter in 1977, who classified the lakes in the north central United States according to their interchange with atmospheric water, surface water, and groundwater variables, lakes had rarely been classified according to multiple hydrologic characteristics and the ones that were categorised, were often separated into the two simple groups: open and closed lakes, defined based on the presence or absence of surface water outlets (Zumberge, 1952; Hutchinson, 1957); outflow and evaporation predominant, defined based on components of the hydrologic budget (Bogoslovsky, 1966); flow-controlled and climate-controlled, defined based on water budget components (Szesztay, 1974); groundwater-dominated and surface water-dominated, defined based on the hydrogeological aspects of lakes (Born et al, 1974). Shannon (1969), Brezonik (1969) and Winter (1977) criticized these simplistic classification approaches, claiming them inadequate and non-transferrable. It should be noted that, although classifications based on a variety of variables are more likely to be of widespread use, their individual limitations should still be acknowledged.

Established in 2000, the European Union (EU) Water Framework Directive (WFD), a continent-wide policy committed to achieving good ecological status of all water bodies by 2015, required member states to identify surface water bodies according to type (UKTAG, 2003). The Water Framework Directive (WFD) recognised that certain hydromorphological lake characteristics (e.g. geology, depth, altitude, size) are important factors in determining the composition and abundance of biological communities within the waterbody (Free et al, 2005). By classifying lakes on a hydromorphological basis, it is possible to distinguish the 'natural' influences of hydrology on the aquatic ecology from 'artificial' influences attributable to anthropogenic impacts and pressures.

Article 2 and Annex II of the WFD demand that each Member State define an initial lake typology with categories differentiated according to hydromorphological type. Ireland and Northern Ireland agreed that System B in Annex II of the WFD would be appropriate to define hydromorphological typology for lakes (UKTAG, 2003; Free et al, 2005). This must be done according to the technical specification set out in Annex II. For example, Ireland and Northern Ireland defined an initial lake reporting typology based on System A of Section 1.2 of Annex II of the Water Framework Directive using a number of typology descriptors. Table 4.10 shows this typology applied to the lake system at Sheskinmore Lough. This classification system defines the open water lake habitat at Sheskinmore Lough as Calcareous (HA) as more than 50% of the geology is composed of calcareous substrates and the conductivity is moderate to high, falling within the range of 151-1000 mS cm⁻¹ as per the edaphic variable monitoring conducted in June 2013. The lake is very shallow (Sh) due to a mean depth of less than 3m, it can be categorised as lowland (Low) due to its location at the coast below 200mOD, and it is small (S) with an open water area of 10-49ha and, as a result, the typology framework suggests it may require monitoring.

The hydrology of the lake over the course of the two-year monitoring period has been classified using a similar system designed by Cui et al (2010), which categorizes each month based on a combination of average monthly water levels and water levels relative to a monthly optimum. As per Cui et al (2010), the monthly optimum was calculated as the median of the difference between the maximum and minimum monthly water levels (Table 4.11). June 2012 was classified with a moderately average monthly water level with less than double the number of sub-optimum days compared to above-optimum days. July 2012 was also classified with a moderately average monthly water level, but in contrast to June, displayed over double the number of sub-optimum days compared to above-optimum days. August 2012 and March 2013 were classified with low average monthly water levels and over double the number of sub-optimum days compared to above-optimum days. September 2012, April 2013 and May 2013 exhibited less than double the number of above-optimum days compared to sub-optimum days. Similarly, October 2012 and February 2013 were classified with moderately average monthly water levels with over double the number of above-optimum days. In contrast, November 2012 was classified with an above average monthly water level with less than double the number of above-optimum days compared to sub-optimum days. December 2012 and January 2013 were also classified with above average monthly water levels, however they displayed over double the number of sub-optimum days compared to above-optimum days.

Table 4.10 Classification of the lake at Sheskinmore Lough using the tiered typology proposed by Northern Ireland and also applied by Ireland to explain biological variation in lakes (Adapted from UKTAG, 2003 and Free et al, 2005).

Tier 1	Feature (Abrev.)	Catchment	Conductivity (mS cm ⁻¹)	Sheskinmore Lough Lake Classification
Geology	Organic (P)	>75% Peat		×
	Siliceous (LA)	>90% Siliceous Solid Geology	<70	×
	Siliceous (MA)	>50% Siliceous Solid Geology	71-150	×
	Calcareous (HA)	>50% Calcareous Geology	151-1000	✓
	Calcareous (Marl)	>65% Limestone	151-1000	×
	Brackish (B)		>1000	×
Tier 2	Feature (Abrev.)		Mean Depth (m)	Sheskinmore Lough Lake Classification
Depth	Very Shallow (Sh)		≤3	✓
	Deep (D)		>3	×
Tier 3	Feature (Abrev.)		Basin Altitude (mOD)	Sheskinmore Lough Lake Classification
Altitude	Lowland (Low)		<200	✓
	Mid-Altitude (Mid)		200-800	×
	High-Altitude (High)		>800	×
Tier 4	Feature (Abrev.)	Operational Definitions	Water Area (ha)	Sheskinmore Lough Lake Classification
Size	Very Small (VS)	Very Small Lakes – only monitored in exceptional circumstances	1-9	×
	Small (S)	Small Lakes which may require monitoring	10-49	✓
	Large (L)	Large Lakes which require monitoring	50-10,000	×

Table 4.11 Hydrological classification of lake water levels based on monthly average and above-optimum/sub-optimum water levels observed during the two-year monitoring period from 18th June 2012 to 17th June 2014 (Cui et al, 2010). Note Class A = >140mm average monthly water level; Class B = 120 -140mm average monthly water level; Class C = <120mm average monthly water level; Class a = Over double the no. of above-optimum days compared to sub-optimum days; Class b = Less than double the no. of above-/sub-optimum days compared to sub-/above-optimum days; Class c = Over double the no. of sub-optimum days compared to above-optimum days.

Indicator	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May
Average Monthly Water Level (mm)	135	137	101	118	132	151	142	151	137	89	100	118
Optimum Monthly Water Level (mm)	136	141	116	115	126	149	146	155	133	89	103	124
Maximum Monthly Water Level (mm)	140	149	139	140	155	163	161	175	175	91	121	158
Minimum Monthly Water Level (mm)	132	132	92	90	96	135	131	135	91	87	85	90
No. of Days per Month Above Monthly Optimum Water Level	4	8	5	19	23	19	8	10	19	10	13	14
No. of Days per Month Below Monthly Optimum Water Level	9	23	26	11	8	11	23	21	9	21	17	17
Class	B, b	B, c	C, c	C, b	B, a	A, b	A, c	A, c	B, a	C, c	C, b	C, b

The distribution of lake flooding frequency ranges are displayed in Figure 4.47, revealing that the open water habitat of the lake is covered by water for over 50% of the two-year survey period. The deeper areas in the centre and northeast corner of the lake are flooded 100% of the time, while the areas to the south and southwest are only covered for half (50%) of the year-long monitoring period. The exception is an area of higher elevation in the centre of the lake where coverage is more intermittent. The southern and western fringes of the lake experience the most infrequent flooding, with areas at the very edge of the open water habitat covered by water for just 25% to 50% of the two-year survey period. When water levels fall below average, these are the areas of the lake that experience the most exposure when water levels fall below average. The distribution also shows that for 25% of the year, flooding occurs beyond the defined edges of the open water habitat of the lake, especially within the reedbed and within the wetland on the southern and western sides of the lake.

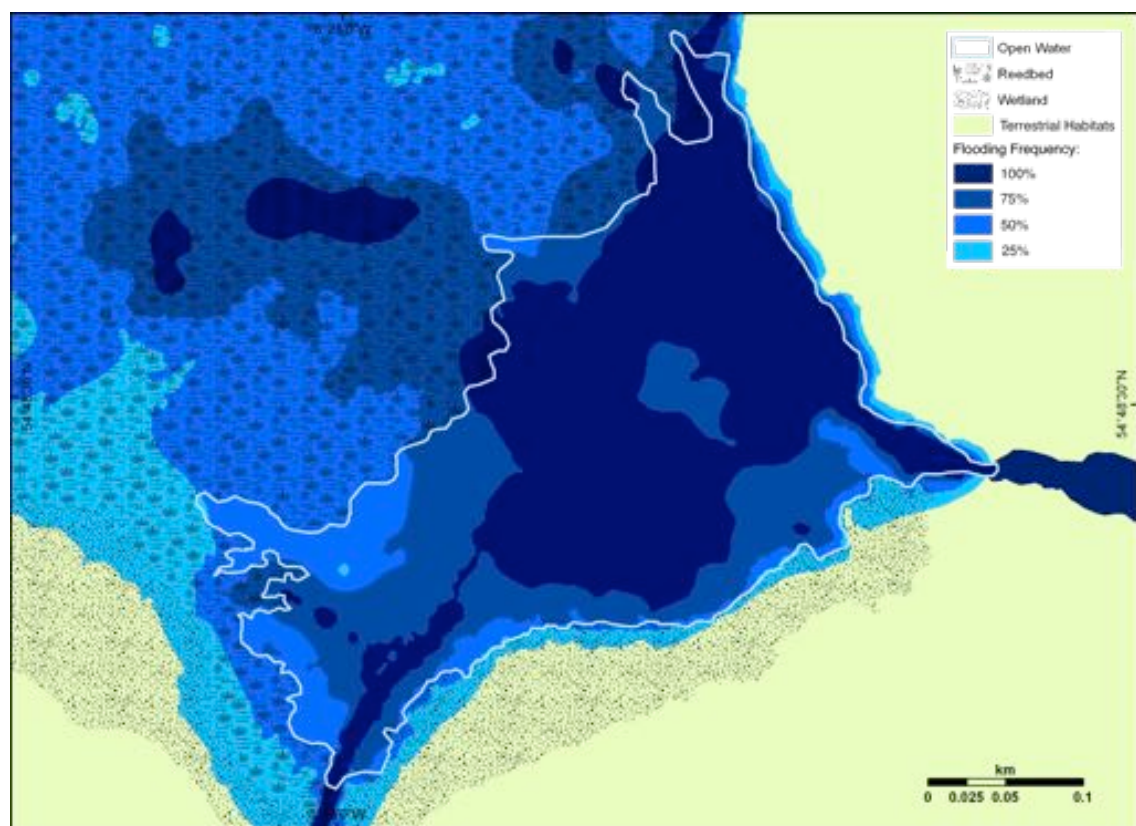


Figure 4.47 Distribution of lake flooding frequency ranges, calculated as the percentage of the two-year survey period (18th June 2012 to 17th June 2014) inundated by more than 10cm of water, relative to the open water habitat of the lake, and the wetland and reedbed systems.

4.5 Discussion

4.5.1 Wetland community-environment relationships

Associations between the edaphic environment and wetland/reedbed communities identified through TWINSpan analysis are examined here using principal component analysis (PCA) of local environmental variables (Figure 4.48). The first two axes of the PCA

represent 23.1% of the species variance in the data, and 87.5% of the species-environment relationships represented. The first axis (17.9% species variance; 68% species-environment association) is strongly negatively correlated with conductivity and total nitrate, and to a lesser extent negatively correlated with soluble reactive phosphorus. The second axis (5.2% species variance; 19.5% species-environment) more closely represents total phosphate and pH. Elevation stands out as a strong gradient despite the small data range, but is not aligned specifically with either of the first two axes. Although the main data gradients are represented by conductivity and nitrate, the division of communities, which almost entirely overlap on the first axis, does not follow these gradients. Instead, the differences between *Fen meadow* and *Peat mire* communities appear to be driven by gradients in pH and total phosphate.

The *Fen meadow* community is most strongly associated with low total phosphate, high soluble reactive phosphate and the full pH range (6.61-7.50). These findings are in accordance with the environmental descriptions provided by the NVC (National Vegetation Classification) and EUR27 (Natura2000) classification systems, from which the wetland and reedbed communities in this study were identified (Elkington et al, 2001; EC, 2007). Specifically, the lowland *Fen meadow* community is described by NVC class M22 as being associated with peat soils that are moist for most of the year due to: 1) their location in, or around, well-developed springs, flushes and mires, or marking out areas of surface or groundwater influence; 2) their moderate to high base-status; and 3) a pH range of 6.5-7.5 (Elkington et al, 2001). The presence of species including *Briza media*, *Leucanthemum vulgare*, *Lotus corniculatus*, *Plantago lanceolata* and *Potentilla erecta*, however, indicate stabilised, base-poor soils more characteristic of calcareous sand dunes. It is therefore no surprise that the community is also associated with moderate conductivities more typical of the EUR27 classification of Irish Machair (class 21A0), which describes such habitats as complex, sandy coastal plains resulting partially from grazing, in an oceanic location with a cool, moist climate (Ritchie, 1975; Bassett & Curtis, 1985; EC, 2007). The wind blown sand has a significant percentage of shell-derived material, forming a lime-rich soil with pH values normally greater than 7 (EC, 2007).

The less extensive first-level *Peat mire* community, on the other hand, is not associated with such low extremes of total phosphate or elevation. Located primarily at the western end of the wetland and at the reedbed fringe, it is instead found at slightly higher elevations in areas with moderate to high total phosphate and high soluble reactive phosphorus concentrations. Like the *Fen meadow* community it occurs across the full wetland and reedbed pH range (6.61-7.50); however it is more strongly associated with lower pH levels and moderate conductivity. Again, this is in accordance with the environmental descriptions provided by the NVC and EUR27 classification systems. Specifically, *Peat mire* is described by NVC class M9 as characteristic of soft, spongy peats kept permanently moist by moderately base-rich and calcareous waters that maintain a pH of 5 and above (Elkington et al, 2001). These descriptions are reflected in the EUR27 classification of Transition Mires and Quaking Bogs (class 7140), which describes such habitats as peat-forming communities developed in the transition zone between oligotrophic to mesotrophic water bodies, with characteristics intermediate between

soligenous and ombrogenous types (Du Rietz, 1949; Grootjans et al, 2006; EC, 2007).

In contrast to the first level communities, the four second-level wetland and reedbed communities identified through TWINSpan analysis are more clearly aligned with some of the environmental gradients evident in the data (Figure 4.48). For example, the *Groundwater mire* community located primarily along the wetland fringe and across the reedbed, is associated with the full range of total phosphate concentrations, average elevation and total nitrate levels. This community shows weak relationships with soluble reactive phosphorus, pH and conductivity, and all can be categorised as moderate. The equivalent NVC community (M10) is described as one of the most calcicolous of mire communities in the British Isles and the pH of the flushing waters is mostly high, usually between 5.5 and 7.0 or sometimes higher. The community is typical of groundwater-fed soligenous mire habitats located on mineral soils and soft, spongy, shallow peats kept wet by base-rich, calcareous and oligotrophic waters (Elkington et al, 2001). Occasionally this community is associated with ombrogenous mires and bogs, and within open water transitions around lakes, hence the EUR27 classification of Blanket Bogs applies here (class 7130), which captures the influence of the surrounding extensive boggy landscapes with poor surface drainage that dominate this area of oceanic northwest Ireland (Doyle & Moore, 1980; EC, 2007).

The more widespread *Sedge mire* community, located around the north and south sides of the lake and the central part of the reedbed, also differs from the other second-level communities in that it is primarily associated with high pH levels and low levels of total nitrate, soluble reactive phosphorus and conductivity, as well as a weak relationship with total phosphate. The equivalent NVC community, *Lowland mire* (M13) is confined to peat or mineral soils, strongly irrigated by base-rich, highly calcareous, and oligotrophic to slightly mesotrophic waters ranging from pH 6.5 to 8 (Elkington et al, 2001). This rush-dominated community is located along groundwater seepage lines, in open water transitions around lakes and beside springs and flushes. The combination of edaphic characteristics and a cool, wet climate exerts a strong influence on the structure and composition of the vegetation in this community. The EUR27 classification of *Molinia* Meadows on Calcareous, Peaty or Clayey-silt-laden Soils applied here (class 6410) also provides an appropriate environmental setting (EC, 2007). Although like *Sedge mire*, this community contains an abundance of *Molinia* spp. and is typically found on wet, nutrient-poor, calcareous soils, they also stem from neutro-alkaline to soils with a fluctuating water table and from the draining of peat bogs (Devillers & Devillers-Terschuren, 1996; Sporek & Rombel-Bryzek, 2005).

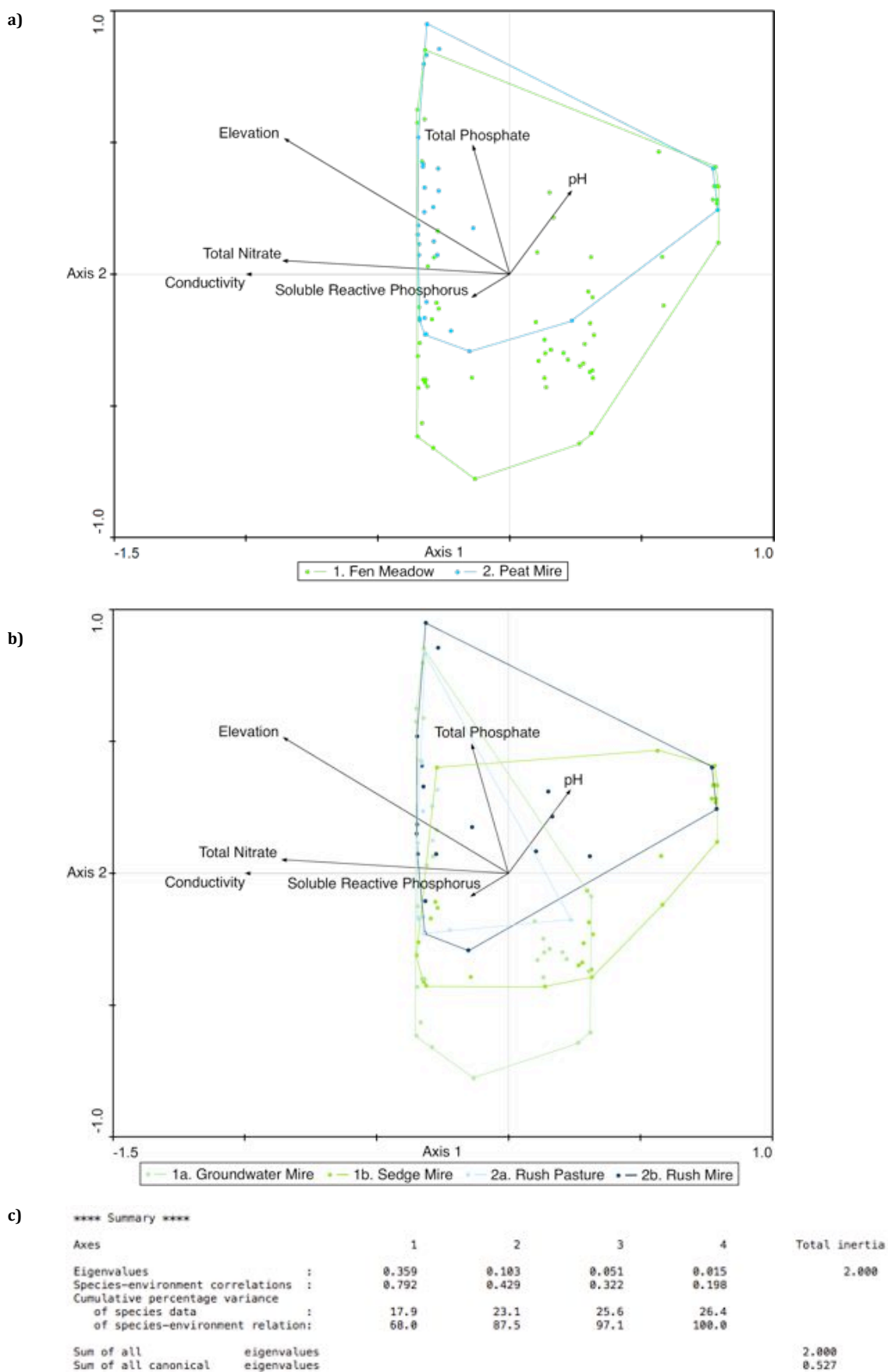


Figure 4.48 PCA showing the relationships between environmental variables measured in June 2013 across the study site and the TWINSpan-defined a) first-level and b) second-level communities of the wetland and reedbed, and c) PCA statistics.

The *Rush pasture* and *Rush mire* communities, which dominate the main wetland arm and fringe the reedbed, are more similar in terms of their environmental associations. Both communities are associated with higher total phosphate levels and pH, as well as high soluble reactive phosphorus concentrations, average total nitrate, conductivity and elevation. The two communities separate more distinctly along the nitrate conductivity and soluble reactive phosphorus gradients, where *Rush mire* extends across the full range of values, whereas *Rush pasture* is more strongly associated with moderate to high levels. There is also a difference in pH with *Rush mire* more strongly associated with higher pH levels. NVC community M28 (*Rush pasture* here) is characteristic of wet, slightly acidic to neutral soils with a pH of 4 to 7, peaty and mineral soils in cool and wet climes. It is an oceanic community of gently-sloping ground that is found in wet hollows and around the margins of soligenous flushes and topogenous mires, as well as on ill-drained, comparatively unimproved or reverted pasture. The EUR27 classification of Alkaline Fens applied here (class 7230) also provides an appropriate environmental setting for the *Rush pasture* community, as it is present on permanently waterlogged peaty soils, with a water supply that is soligenous or topogenous base-rich, and can often be calcareous (EC, 2007). Similarly, NVC community M23 describes the second-level *Rush mire* community as confined to peat or mineral soils strongly irrigated by base-rich, highly calcareous, and oligotrophic waters ranging from pH 6.5 to 8. This rush-dominated community is located along groundwater seepage lines, in open water transitions around lakes and beside springs and flushes. The environmental setting provided by the EUR27 classification of Oligotrophic to Mesotrophic Standing Waters applied here (class 3130) also supports these findings as it represents short perennial vegetation on nutrient-poor soils along land-water interface zones of oligotrophic to mesotrophic lakes, pools and ponds which experience periodic drying (Jenssen, 1979; EC, 2007).

Within Sheskinmore Lough itself, edaphic factors have also been explored collectively to ascertain the likely environmental influences on the submerged macrophyte communities. Table 4.12 details the hydrochemical environment of Sheskinmore and Sandfield Lough in the context of Irish (Republic of Ireland and Northern Ireland) and European hydrochemical ranges observed across a total of 860 lakes. Both lakes lie well within the ranges nationally and across Europe. Sheskinmore Lough, with a slightly acidic mean pH of 6.38 matches the mean pH observed within the 86 lakes surveyed in Europe and is close to the mean pH of 6.1 observed within the 200 lakes in the Republic of Ireland. In contrast, the mean for Northern Ireland is slightly more alkaline at pH 7.69, which is closer to the neutral pH of 7.02 observed within Sandfield Lough. These findings are also in accordance with the environmental descriptions provided by the NVC and EUR27 classification systems, from which the lake communities in this study were also identified (Elkington et al, 2001; EC, 2007). Specifically, all of the communities identified by NVC and EUR27 classifications (E, 3130: *Littoral coastal*; E, 3140: *Benthic coastal*; C2, 3110: *Benthic lowland*; E, I, 3140: *Shallow coastal*; B, 3140: *Shallow lowland*) are described as water bodies that contain a circumneutral pH. The exception is *Littoral lowland* community (I, 1130), which is characterised by a moderate alkalinity.

The mean conductivity level recorded at Sheskinmore Lough ($180 \mu\text{S cm}^{-1}$) falls well below the mean values observed within Northern Ireland ($255 \mu\text{S cm}^{-1}$) and across Europe ($978 \mu\text{S cm}^{-1}$). Sandfield Lough has higher conductivity levels ($346 \mu\text{S cm}^{-1}$), exceeding the mean for Northern Ireland but still moderate. The environmental descriptions provided by the NVC and EUR27 classified communities support the findings for both lakes, with low (C2, 3110: *Benthic coastal*; B, 3140: *Shallow lowland*) to moderate (E, 3130: *Littoral coastal*; E, 3140: *Shallow coastal*; I, 1130: *Littoral lowland*; E, I, 3140: *Shallow coastal*) conductivity (Elkington et al, 2001; EC, 2007).

Table 4.12 The hydrochemical environment of Sheskinmore Lough and Sandfield Lough (recorded in June 2013) in the context of Irish and European lake hydrochemical ranges observed from three studies that have surveyed the environment of 774 Irish lakes (200 lakes in the Republic of Ireland (Aherne et al, 2002) 574 lakes in Northern Ireland (Heegaard et al, 2001)), and 86 lakes in Europe (Nöges et al, 2003).

Hydrochemical Variable	Sheskinmore Lough (L3) Mean	Sandfield Lough (L6) Mean	Minimum	Mean	Maximum	Region	Source
pH	6.38	7.02	4.10	6.10	8.50	Ireland	Aherne et al, 2002
			4.02	7.69	9.91	Northern Ireland	Heegaard et al, 2001
			4.28	6.38	9.93	Europe	Nöges et al, 2003
Conductivity ($\mu\text{S cm}^{-1}$)	180.03	346.33	38	255	670	Northern Ireland	Heegaard et al, 2001
			5	978	1890	Europe	Nöges et al, 2003
Total Nitrate (mgL^{-1})	0.148	0.150	0	0.13	0.33	Ireland	Aherne et al, 2002
			0	0.10	0.20	Northern Ireland	Heegaard et al, 2001
			0.02	0.19	0.29	Europe	Nöges et al, 2003
Total Phosphate (mgL^{-1})	0.010	0.021	0.002	0.098	0.753	Europe	Nöges et al, 2003
Soluble Reactive Phosphorus (mgL^{-1})	0.002	0.004	0	0.004	0.007	Northern Ireland	Heegaard et al, 2001

The mean total nitrate level observed within Sheskinmore Lough (0.148 mgL^{-1}) is virtually identical to Sandfield Lough (0.15 mgL^{-1}) and falls above the mean levels of Ireland (0.13 mgL^{-1}) and Northern Ireland (0.1 mgL^{-1}). This is in contrast to European lakes, which contain comparatively higher mean total nitrate levels (0.19 mgL^{-1}). Mean total phosphate levels within Sheskinmore Lough (0.01 mgL^{-1}) are comparable to the European mean (0.098 mgL^{-1}), while the Sandfield Lough displays a slightly higher level at 0.021 mgL^{-1} . Mean soluble reactive phosphorus levels within Sheskinmore Lough (0.002 mgL^{-1}) are lower than the Northern Ireland mean of 0.004 mgL^{-1} , whereas Sandfield Lough is the same (0.004 mgL^{-1}). The low nutrient levels observed in both lakes are reflected in the NVC and EUR27 community classifications, which are all characterised as oligotrophic to slightly mesotrophic standing waters (Elkington et al, 2001; EC, 2007).

Analyses of the relationships between the ecological and environmental conditions have revealed three main abiotic controls on the species and communities within the contemporary lake and wetland system at Sheskinmore Lough: physical context, geology and chemistry. These three factors do not impact the ecology of the site in isolation however, in combination they influence the diversity, distribution and types of communities found across the lake and wetland system. In addition, they also provide a means by which to set the scene for ecohydrological analyses. For example, after climate,

the physical context is the overriding factor determining lake and wetland community diversity, distribution and type, as it fundamentally determines water supply and water quality characteristics (Humphries & Meade, 2007). Similarly, geology and soils also determine community characteristic as they influence the hydrological conditions across a site, and also the chemistry of the soil substrates thereby determining botanical composition and character of aquatic communities.

The physical context is a key contributing factor to the number and diversity of communities identified across the site (Humphries & Meade, 2007). Here, the close proximity of the system to the coast provides a unique range of physical stressors including saline influences, from sea spray during the higher wind speed conditions. Saline inputs are also possible via surface inflow and groundwater intrusion, but these usually occur in association with high energy (event-driven) overwash processes, or associated with long-term below ground movements (DeLaune et al, 1987). Although short-term events such as storms are an important influence, the array of freshwater communities identified and the low conductivities measured across the site suggest that salinity is not particularly important across Sheskinmore. In contrast, calcareous inputs in the form of shell material, which originate from significant geomorphological shifts in the surrounding sand dunes driven by strong aeolian forces, have a strong contemporary influence on community assemblage locally (Rodwell, 1985; Barrett-Mold & Burningham, 2010; Barrett-Mold, 2012). This is corroborated here in that aeolian-transported shell inputs to the system have promoted calcareous conditions helping to shape the first-level wetland community, *Peat mire*, and also the second-level communities *Groundwater mire*, *Sedge mire* and *Rush mire*; these are all characterised by species assemblages that favour soils irrigated by highly calcareous waters (Elkington et al, 2001; Hill et al, 1999).

The physical structure of the lake at the confluence of the Abberachrin and Duvoge rivers is also a key contributing factor to the distribution of communities observed within the lake. De Paggi & Paggi (2008) and Lesack & Marsh (2010) observed that river-to-lake connectivity is an important mechanism driving enhanced biodiversity in lacustrine systems, in contrast to those lake systems with no riverine influence. The highest submerged macrophyte species abundance and community diversity is found around the river-lake junctions, and also along the channel feature within the lake. River-to-lake connectivity provides an important source of sediments and nutrients to an oligotrophic lake that are otherwise dominated by sandy substrates (De Paggi & Paggi, 2008; Lesack & Marsh, 2010). Within Sheskinmore Lough, the increased abundance and community diversity might be explained by the presence of fine riverine organic silt lining the sides of the channel, providing a favourable substrate and anchorage to support increased macrophyte productivity (Abrahams, 2008). The channel also provides a more sheltered and deeper environment, reducing disturbance impact from waves and associated subsurface turbulence and erosion (De Paggi & Paggi, 2008).

The biodiverse and silty channel is in contrast to the flat shallow areas of the lake, which are comparatively more exposed to the erosion effects of wind-waves, which facilitate sand-dominated substrates. In these areas, only low-growing, resilient, pioneer

macrophyte species can survive. Charophytes, such as *Chara aspera*, do well here as they are often the first macrophytes to colonize a water body after disturbance. They are typical of shallow water environments and are distinct, not just taxonomically, but ecologically to other macrophytes. In some lakes, distinct shifts from diverse macrophyte communities to an almost complete monoculture of charophytes has been observed (Van den Berg et al, 1999; Van den Berg et al, 2001). This suggests the co-existence of these two groups in the lake may be unstable, with disturbance potentially leading to alternative stable states of dominance by either of the groups (Van Nes et al, 2002). For example, once it has colonised an area of lake bed, *Chara aspera* will deplete the substrate of bicarbonate concentrations preventing other macrophyte species from colonising (Van den Berg, 1999). Thus there is a positive feedback in the development of *Chara aspera* as it drives the system towards carbon limitation and, via interspecific competition, suppresses the establishment of other macrophytes and in turn creates better light conditions for itself (Van Nes et al, 2003). In comparison, macrophytes in the deeper channels of the lake thrive in these areas due to the continual replenishment of river-borne silty sediments. The organic substrate not only supports their establishment, it is abundant enough to ensure their populations are maintained and they dominate over *Chara aspera* due to their superior competitive ability for light (Van den Berg et al, 2001). These findings show that subtle variations in sediment distribution across the lakebed are strong enough to drive the overall contemporary spatial community ecology patterns within Sheskinmore Lough.

Physical characteristics of the local topography also imprint the ecohydrology of the lake and wetland. The steep impermeable granitic topography to the north promotes overland flow during precipitation events, evidenced by the flashy water level regime recorded at dipwells at the base of this hinterland. Winter (2000) observed that hydrologic landscapes are defined by the flow characteristics of groundwater and surface water and by the interaction of atmospheric water, surface water, and groundwater for any given locality or region. Groundwater flow therefore has an important part to play in determining site ecology, a process that is strongly influenced by landscape topography. Where overland flow is minimal in the sand dune system to the south and west, groundwater flow dominates, producing the hydrological conditions that favour the establishment of *Fen meadow*, *Groundwater mire* and *Rush mire* communities which are associated with springs and groundwater seepage lines (Elkington et al, 2001). Topography, and its influence on regional climate and hydrology, is therefore a key factor contributing to the community assemblage patterns observed across the lake and wetland system at Sheskinmore Lough.

Surficial geology across the Sheskinmore site contrasts between peat-dominated uplands to the north and east, and coastal sand dunes to the south and west, which both overlay granitic bedrock. Halliday Traut (2005) observed that coastal ecotones harbour increased community diversity, and biogeochemical processes are important for determining vegetation distribution. Dickinson & Mark (1994) observed that coastal dune to peat wetland ecotones increased in community diversity with distance from the coast. At Sheskinmore Lough, the community pattern is more complicated, however, as the ecotone is not uniform across the site. This is largely due to the shape of the coastline and Loughros More valley, which promotes uneven substrate distribution as a result of varied

aeolian sand inputs that have centres of mass to the south and west of the lake and wetland (Barrett-Mold, 2012). This is reflected in the varied distribution of the *Groundwater mire* community, which has stronger associations with non-peat substrates and is found across the site, from the wetland periphery to the centre of the reedbed (JNCC, 2015). Although *Groundwater mire*, and all of the communities identified within the wetland and reedbed system are associated with peat substrates, the geological mosaic of the dune-peat transition zone across the site contributes more subtle variations in community distribution. According to Rodwell (1985), the transition zone from wetland communities to coastal sand dune communities is one that is usually uneven, and the intermixture of dune communities within a wetland complex, and vice versa, is particularly noticeable. At Sheskinmore Lough, the *Rush pasture* (M28) and *Rush mire* (M23) communities represent this interrupted landward transition from dune complex to wetland habitat, which stretches across the lake and wetland system.

To some extent, these physical and geological factors also drive variations in chemical influence on the communities identified across the lake and wetland system. In the areas towards the western extent (seaward, dune-covered) of the wetland, alkaline pH levels dominate and conductivity levels are high due to the calcareous nature of the substrate. At this end of the site, the dominant macrophyte assemblages include *Peat mire* and *Rush mire* communities, which are most tolerant of alkaline, high conductivity conditions. Conversely, the northern and eastern extents (bedrock and blanket bog hinterlands) of the site have lower pH levels and the proliferation of communities, including *Groundwater mire* and *Rush pasture*, contain species that are more tolerant of lower pH levels (Hill et al, 1999; Elkington et al, 2001). Although the availability of the nutrients (total nitrate, total phosphate and soluble reactive phosphorus) usually determines plant growth rate, competitive advantage and the structure of plant communities in aquatic ecosystems (phosphorus, as phosphate, is probably the more limiting in semi-natural oligotrophic systems), the levels recorded across the site were so low they are unlikely to have a distinct impact (Humphries & Meade, 2007).

This study has highlighted physical context, geology and chemistry as the three key environmental factors primarily influencing the distribution, diversity and types of aquatic communities. These edaphic factors provide the suite of abiotic conditions, within the lake and throughout the wetland and reedbed systems, responsible for producing a patchwork of ecological niches, which currently support the array of contemporary communities identified at Sheskinmore Lough during this study.

4.5.2 Hydrological requirements of communities and habitats

Hydrological controls on the ecology across the site are best evaluated using the Sum Exceedence Value (SEV) concept, defined by Gowing et al (2002) and based on the earlier Dutch work by Sieben (1965). This method requires threshold depths being specified for hydrological monitoring sites: one defines the water table depth at which the zone of densest rooting (taken to be 0-100mm depth) begins to become waterlogged, the other defines when drying of the surface soil becomes detectable by plants. The waterlogging

threshold is calculated from a soil moisture release curve as the depth that gives 10% air-filled porosity, while the soil drying threshold is calculated using the Richard's equation as the depth that gives 0.5 m tension at the surface (Gardner, 1958; Gowling et al, 2002). The advantage of using the SEV approach with site-specific hydrological regimes is that the resultant information is transferable between sites (Gowling et al, 2002). Data from all of the sites can be combined to show the full range of hydrological regimes across the wetland, and in turn this facilitates direct comparison with ecological data.

The SEV soil waterlogging and SEV soil drying was calculated at each of the hydrological monitoring sites across the wetland that were active during the first full year of monitoring (18th June 2012 to 17th June 2013), specifically W1-3 and W5-7 (Figure 4.49). At the far west of the wetland (W1) and in the area just to the west of the lake (W3), water levels regularly fluctuate driving distinct shifts from waterlogged to dry conditions during the summer period. From a waterlogged baseline at c. 35mm below the ground surface (June to October 2012) there are sharp drops in water level leading to episodes of drying in July, August and early September. Following waterlogged conditions in early spring, a drying episode occurs in April and June 2013. At site W2, in the central arm of the wetland, dry conditions prevail for 230 days in total, experiencing no soil waterlogging conditions (outside the October to February period of vegetative inactivity). Nevertheless, W2 displays a similar water level pattern to W1 and W3 throughout the year.

The soil at W5 (at the base of the valley side, to the north of the lake) is permanently waterlogged and water levels exceed the ground surface throughout the majority of the autumn and winter. The closest the water levels at this site come to falling below the SEV soil waterlogging threshold is on 11th August 2012 and 6th April 2013 when water levels fall to just 10mm and 20mm below the ground surface, respectively. During the periods of soil waterlogging in 2012 and 2013 there are three peaks in water level: 5th July 2012, 13th August 2012 and 2nd May 2013. W5 does not display the short periods of spring and summer drying observed elsewhere, although it does mirror these very slightly in pattern. Sites W6 and W7, located along the south margin of the wetland, exhibit more pronounced fluctuations between soil waterlogging and drying in comparison to the pattern observed at W1-W3. Waterlogging occurs for longer periods than drying, dominating the spring, summer and early autumn period. W7 differs from W6, however, in that water rises above the ground surface, whereas at W6 the ground surface is never breached. The peaks in water level observed at W6 and W7 in late January and early February 2013, are accentuated at W5, where water levels increase to 185mm above the ground surface.

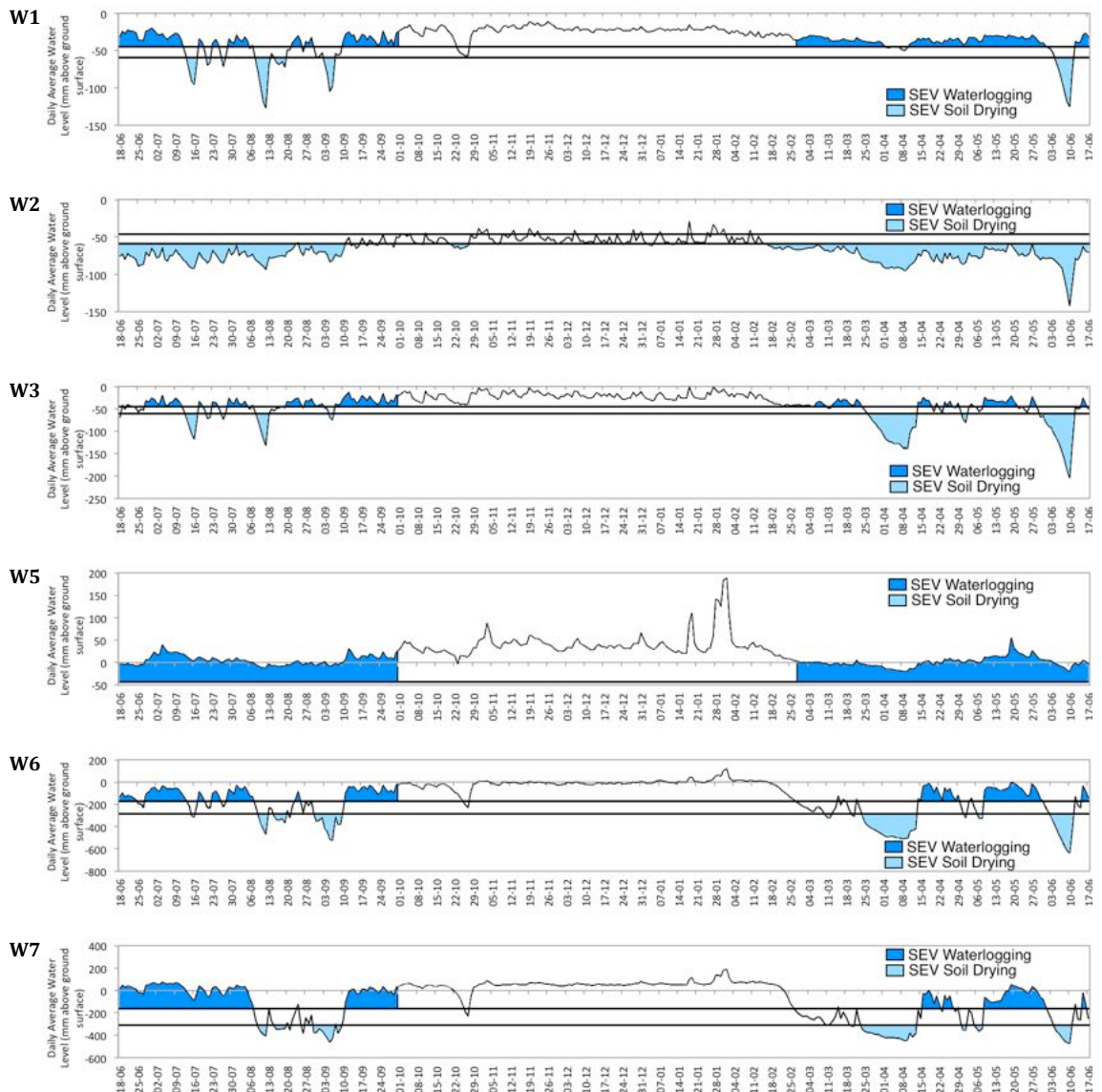


Figure 4.49 SEV soil waterlogging (dark blue shaded area) and SEV soil drying (light blue shaded area) calculated from 18th June 2012 to 17th June 2013 at the hydrological monitoring sites in the wetland: W1-3; and W5-7. The horizontal lines represent threshold depths for the particular soil type (-48mm to -55mm for peat; -180mm to -320mm for sandy peat): soil waterlogging threshold (upper line); soil drying threshold (lower line). Note the absence of soil waterlogging from October to February as it is only cumulated during the period of active vegetation growth (March to September), when plants are most sensitive to the oxygen status in their root zone (Gowling et al, 2002). Note also the varying vertical scales.

When combined, the SEV soil waterlogging drying data from all of the wetland sites reveal the total spread of hydrological regimes across the wetland system (Figure 4.50). The location of south lake-side sites W6 and W7 to the centre right of the graph indicates greatly fluctuating water regimes between waterlogged and dry states. The hydrological regimes at W1 and W3 (in the main western arm of the wetland) are more stable than W6 and W7; however these sites do still fluctuate between waterlogged and dry soil conditions to a significant degree, and the soil at W1 is more waterlogged than W3. In contrast, the water regime at W2 is constantly dry and at W5 permanently waterlogged.

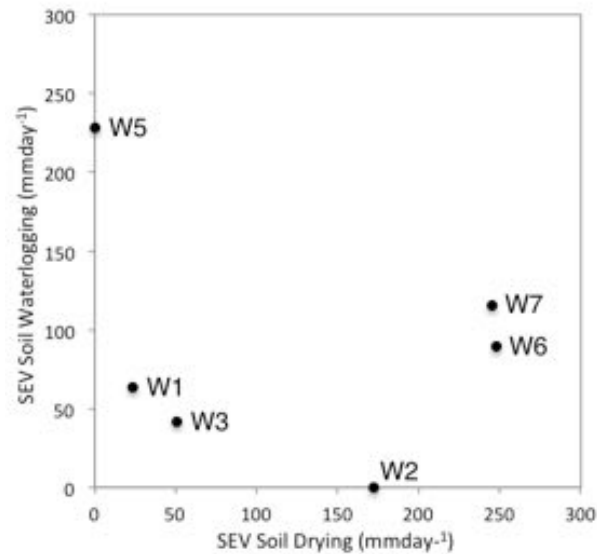


Figure 4.50 Water regime for all of the 6 wetland hydrological monitoring sites as defined by their SEV waterlogging and SEV soil drying calculated as mm per day, recorded from 18th June 2012 to 17th June 2013.

Direct comparison of ecological data collected during the surveys of June 2012 / 2013 and the hydrological regimes identified using the SEV approach was undertaken. The SEV soil waterlogging and drying variables were calculated for the 120 wetland and reedbed quadrats. The water regimes for each of the two TWINSpan-defined first-level communities and each of the four second-level communities were determined by associating each quadrat, and its community type, with one of the six hydrological monitoring sites through interpolation (Figure 4.51). The *Fen meadow* community is associated with water levels that fluctuate more significantly than those attributed to areas dominated by the *Peat mire* community. As a result, it is not related to a distinct water regime, but is characterised by the full spectrum of hydrological variation, including the upper extremes of soil waterlogging and soil drying. This is partly unsurprising given the NVC definition of the community, which describes its hydrological niche as non-habitat specific. Instead, it is typically found either in, or around, well-developed springs, flushes and mires, marking out areas of surface or groundwater influence that cause soils to be moist for most of the year (Elkington et al, 2001).

The *Fen meadow* community is suited to above average levels of waterlogging. It is not, however, suited to high levels of soil drying (Mountford et al, 2006). Instead, low water levels are described by Wheeler et al (2009) to adversely affect the community, so much so that dry conditions tend to prompt the loss of species. The comparatively highly variable nature of the water level regimes at sites W6 and W7 located next to the lake, suggests that lake water is having a direct impact on surrounding hydrology and water regime. If extreme soil drying events become too frequent and intense, the wetland *Fen meadow* community could potentially decline in structure and extent. For instance, frequent and sustained periods of soil drying will have a greater impact on species with low tolerance to dry conditions. Although the majority of *Fen meadow* species would survive a period of drying as many wetland plants tolerate water stress due to presence of in-soil seed banks, their ability to recolonise during frequent prolonged dry periods is

likely to be compromised by displacement from species more suited to drier conditions (Keddy & Reznicek, 1982; Keddy & Constabel, 1986; Wilson & Keddy, 1986; Fojt 1994; Liua et al, 2006; Herrera-Pantoja et al, 2012).

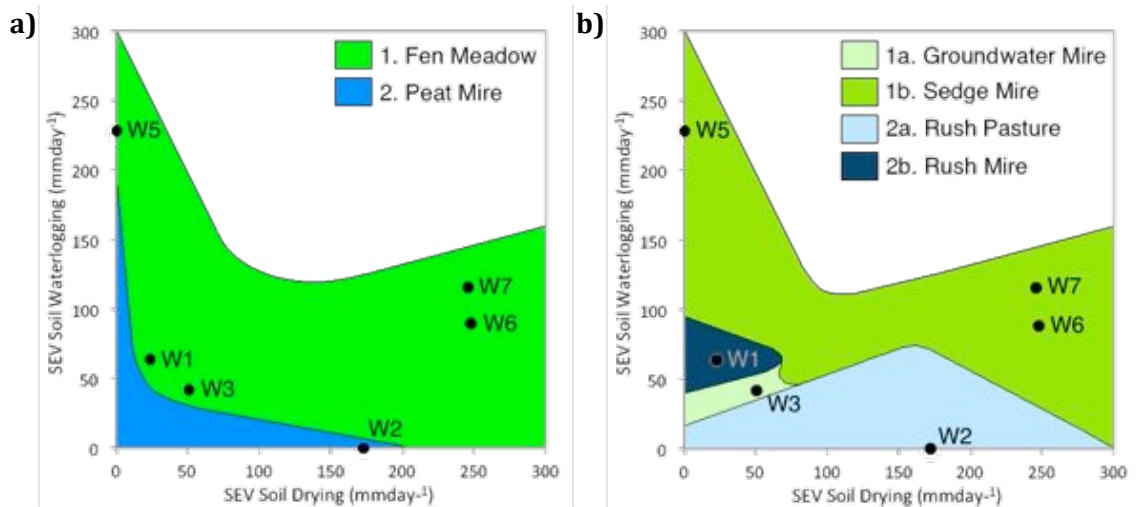


Figure 4.51 Water regime of each TWINSpan-defined community a) first-level communities and b) second-level communities, based on interpolation between the 120 wetland and reedbed vegetation survey sites and the 6 wetland hydrological monitoring sites (W1-3 and W5-7) as defined by their SEV waterlogging and SEV soil drying calculated as mm per day, recorded from 18th June 2012 to 17th July 2013.

In contrast to the *Fen meadow*, the *Peat mire* water regime is more limited and is confined to a moderate range of soil waterlogging and soil drying. It is located in areas of moderately fluctuating water levels, associated with areas of the wetland near W1 and W5, which experience moderate soil waterlogging and very little soil drying. The NVC account for community M9 describes soft, spongy peats kept permanently moist by calcareous waters, indicative of both surface water-fed topogenous and groundwater-fed soligenous mires, and is commonest in wetter areas, often located in natural hollows or old peat-workings, as well as around springs (Elkington et al, 2001). Conversely, it is also found in areas in the centre of the wetland that experience moderate levels of soil dryness and experience little (W3) or no (W2) waterlogging. Interestingly, the second-level *Groundwater mire* community is also associated with the fluctuating hydrology at site W3, suggesting a degree of fluctuation occurs in this part of the wetland, between very low levels of soil waterlogging and the lowest levels of soil drying. This is in accordance with the NVC description for community M10, which typifies the community as one that is groundwater-fed and often found associated with spring and rill vegetation within grasslands and more occasionally within ombrogenous mires, as well as around topogenous mires and within open water transitions around lakes (Elkington et al, 2001).

In contrast to W3, W6 and W7, site W2 experiences no waterlogging. This site is deceiving, however, as it is situated in the middle of a waterlogged area but close to an old man-made elevated field boundary. This *Rush pasture* community is found where soil waterlogging is low or absent. Although it occurs at any level of soil drying, the community is most strongly associated with areas exhibiting moderate levels of soil drying combined with low levels of soil waterlogging. This community (NVC M28) is found on soils that are kept moist to wet for most of the year, around the margins of soligenous flushes and

topogenous mires, as well as on ill-drained, comparatively unimproved or reverted pasture (Elkington et al, 2001). Site W5 is permanently waterlogged; it is located close to the edge of the flooded area of the reedbed and is associated with several communities including *Sedge mire*. But the hydrology here is influenced strongly by overland flow from the upland area to the north of the lake, which precludes soil drying despite the significant fluctuations in water level within the lake close by.

Sedge mire is characterised by the extremes of both high levels of soil waterlogging and high levels of soil drying, although its NVC equivalent (M13) is described as locating on soils where waterlogging is highest and drying is low (Elkington et al, 2001). *Sedge mire* clearly requires a certain level of waterlogging, around at least 100mmday⁻¹, as it shows no association with low levels of soil waterlogging. Despite suitable hydrological conditions to support *Peat mire* species, the *Fen meadow* and *Sedge mire* communities dominate, as waterlogging is restricted to such a limited area that *Peat mire* species cannot establish in significant densities (Squires & Van der Valk, 1992). The *Rush mire* community has a more niche hydrological regime (based on NVC M23) associated with oligotrophic waters located along groundwater seepage lines, in open water transitions around lakes and beside springs and flushes (Elkington et al, 2001). *Rush mire* is associated specifically with moderate to low levels of soil waterlogging, but a near absence of soil drying. This second-level community is absent from areas where there is no soil waterlogging and is most strongly associated with W1 at the far western end of the wetland.

To further assess the contemporary ecohydrology of the lake and wetland system, the macrophyte zonation was illustrated in the context of water levels (Figure 4.52). The diagrams display the relationship between the submerged macrophyte species along transects within the lake and the depth of the minimum, average and maximum monthly water levels defined using the system designed by Cui et al (2010). In the northern, and deepest, section of the lake the base of the channel (1.2m depth) contained species including *Myriophyllum alterniflorum*, *Najas flexilis*, *Nitella flexilis* and *N. translucens* and *Sparganium angustifolium*. Apart from *Myriophyllum alterniflorum*, the base of the southwestern channel section also contains these species. These species are abundant within the deepest section of the channel feature most likely due to the presence of silt. For example, *Nitella translucens* grows primarily on silt substrates, and is rarely associated with sand (Hriynak et al, 2001). Similarly, *Nitella flexilis* prefers the silted areas of lake inlets and *Sparganium angustifolium* favours fine organic substrates (Haslam et al, 1975; Nygaard & Sand-Jensen, 1981; Lucassen et al, 2009). The abundance of *Myriophyllum alterniflorum* and *Najas flexilis* in the channel is more likely to be determined by bathymetry. Although found in shallow waters, and on organic as well as sandy substrates, *Myriophyllum alterniflorum*, for instance, is more commonly associated with moderate depths of 1m to 3m, while *Najas flexilis* is most common in deeper lakes, as it is most typically found in depths of between 1m and 9m (Sheldon & Boylen, 1977; Sarika-Hatzinikolaou et al, 1994). It is also worth noting that *Nitella flexilis* and *Sparganium angustifolium* favour deeper areas of lakes as they are generally more sheltered (Haslam et al, 1975; Nygaard & Sand-Jensen, 1981; Lucassen et al, 2009).

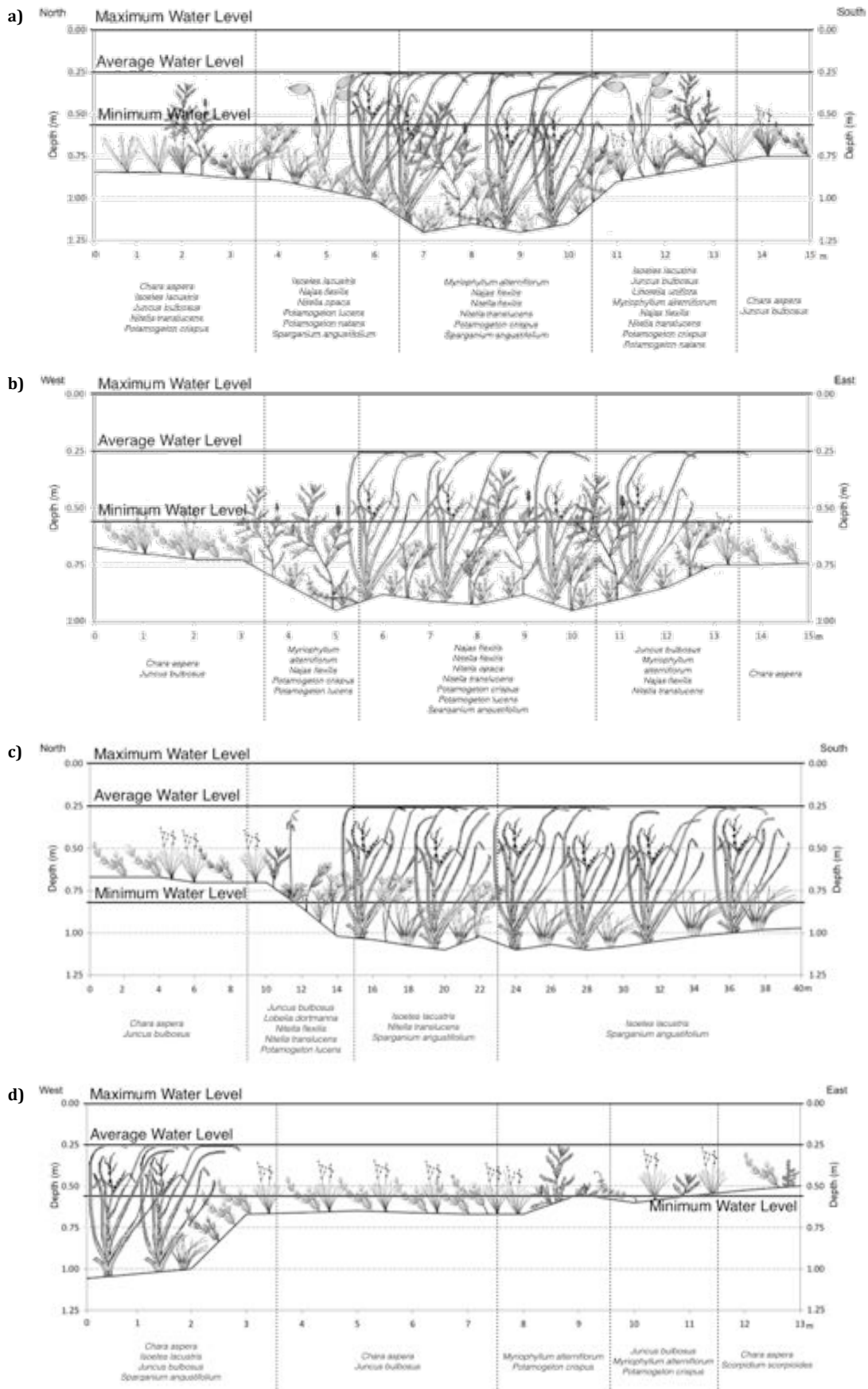


Figure 4.52 Schematic of submerged macrophyte species identified on 14th July 2014 along the four transects across the a) northern and b) southern channel feature, and at the c) southern and d) north-eastern margins within the open water habitat of the lake, in relation to the depth of the minimum and average monthly water levels relative to the maximum monthly water levels defined using the system designed by Cui et al (2010).

The maximum depth range observed within Sheskinmore Lough (0-1.2m) falls at the lower end of the depth ranges preferred by the majority of the species identified within the lake (Figure 4.53), including: *Lobelia dortmanna* (1-3m); *Najas flexilis* (1-9m); *Nitella flexilis* (1-10m); *Potamogeton crispus* (1-3m); and *P. natans* (1-2m). In contrast, the water level range exceeds the whole preferred range of *Chara aspera* (0.05-0.6m) and encompasses the majority of the preferred depth ranges of *Nitella translucens* (0.1-2m) and *Sparganium angustifolium* (0.3-1.5m). In the deep area of the northern channel section, the water level decreases to around 0.6m depth in the deepest areas, and 0.3m to 0.5m on the sloping sides. Within the southwestern channel section, minimum water levels reduced the depth of water in shallow areas to just 0.2m and 0.25-0.0m in moderate depth areas, almost exposing species including *Juncus bulbosus* and *Myriophyllum alterniflorum*. At the base of the channel section in this part of the lake, water levels lower to a minimum of 0.4m, reducing coverage for species such as *Nitella flexilis* and *Potamogeton lucens*. When at their maximum, however, water levels produced depths of 0.75m in the shallow areas either side of the channel, and depths of up to 0.9m within the channel itself.

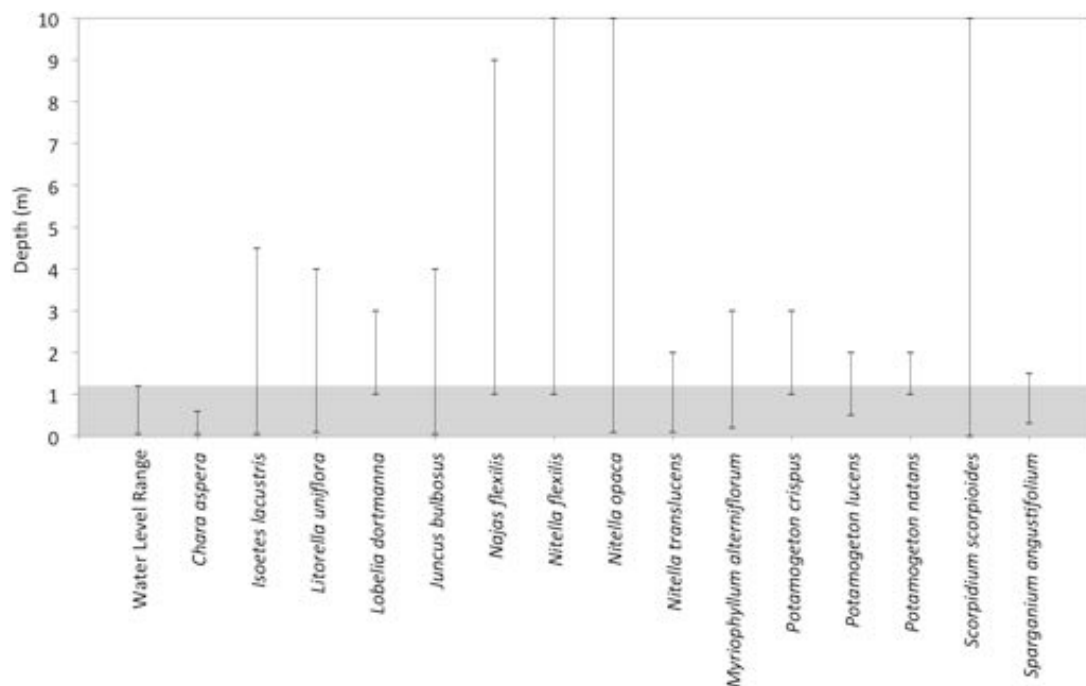


Figure 4.53 Maximum water depth range observed across the entire Sheskinmore Lough open water area during the survey period relative to the preferred water depths of each of the submerged macrophyte species identified in June 2012 and July 2013. Note the dark grey shaded area which has been included to aid comparison.

A number of species, absent from the deepest section of the channel transects, were found growing in moderate depths (0.7-1m) of the sloping channel sides and in the shallower areas outside of the channel. These included *Chara aspera*, *Isoetes lacustris*, *Juncus bulbosus*, *Littorella uniflora*, *Nitella opaca*, *Potamogeton lucens*, *P. crispus* and *P. natans*. *Chara aspera* favours shallower waters, but can be found up to depths of several metres; therefore its presence either side of the channel at depths of 0.75m to 0.85m is to be expected and it is likely that it is outcompeted for light within the channel itself (Van den

Berg et al, 2001). It is found in the shallow areas throughout the lakebed area as it is adapted to nutrient-poor sandy substrates and outcompetes other species by reducing the bicarbonate content of the substrate. *Chara aspera* is also the first to colonise when water levels fall to their minimum levels in the moderate to shallowest areas (Van den Berg, 2001). The water level range observed during the survey period fell to just 0.3m in the shallow areas either side of the channel transects, a depth that would still support *Chara aspera*, *Juncus bulbosus* and other low-growing species. When water levels reach their maximum, however, these areas were covered by 0.8m of water on average. The presence of *Isoetes lacustris* on the northern side of the channel is primarily due to the shelter the deep channel provides from southwesterly wave disturbance, whilst its presence in the shallow area outside of the channel can be attributed to the protection provided by the taller species within the channel (Szmeja, 1994). As its name suggests, *Littorella uniflora* favours the shallower depths of the littoral zone and it is common around the margins of slow moving rivers (Szmeja, 1994). Given the channel feature still exists within the lake, the presence of *Littorella uniflora* here may indicate continued preferential flow of water through the in-lake channel system. Similarly, *Potamogeton lucens* is found in still or slow-flowing water, and although it favours deeper water, it can persist in shallow areas, however it is intolerant of major changes in water levels (Vestergaard & Sand-Jensen, 2000; Van Geest et al, 2005).

At the southern margin of the lake, minimum water levels resulted in exposure of species in the shallows where species such as *Myriophyllum alterniflorum* and *Potamogeton crispus* are abundant, while the majority of the transect was covered in only 0.1-0.15m of water where *Chara aspera* and *Juncus bulbosus* dominate. The deeper areas, dominated by *Sparganium angustifolium*, were covered by 0.5m of water when water levels were at their minimum. When at their maximum, water depths in the shallows exceed 0.5m and in the deeper areas were over 1m. Similarly, the northwestern margin of the lake, dominated by *Chara aspera* and *Juncus bulbosus*, became exposed when water levels were at their minimum and in the deeper areas, abundant with species including *Isoetes lacustris* and *Sparganium angustifolium*, were covered by less than 0.25m water depth. When at their maximum, however, water levels on this side of the lake exceeded 1.1m. The margins of the lake have therefore experienced the longest periods of exposure during the survey period, an observation which can be assessed in a spatial context (Figure 4.54), to illustrate the distribution of lakebed hydroperiod and exposure (covered by < 10cm). The most frequently exposed areas (bare for 50% to 75% of the year) are confined to the very edge of the open water habitat of the lake. During periods of low water levels, species in the littoral zone here will be most impacted, including *Chara aspera* and *Juncus bulbosus*. These species, however, are the most tolerant of exposure and are often the first to recolonise such areas when water levels rise again (Brandrud & Roelefs, 1995; Van den Berg, 2001). The areas exposed for 25% of the year are more widespread, however, dominating the southern half of the lake, as well as a wide swath of the northern area, where they stretch into the centre, forming an island feature during periods of minimum water levels.

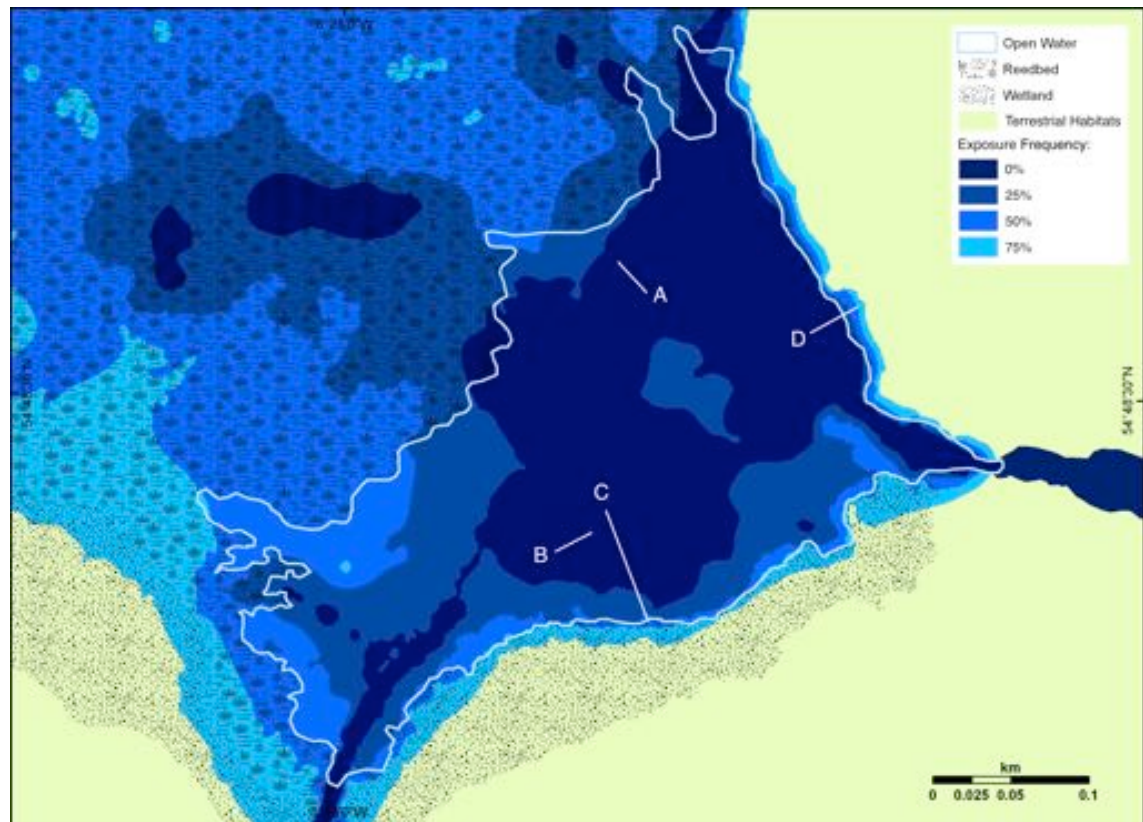


Figure 4.54 Distribution of exposure frequencies, calculated as the percentage of the survey period (18th June 2012 to 17th June 2013) inundated by less than 10cm of water, relative to the four transects (within the A) northern and B) southern channel feature, and on the C) southern and D) north-eastern margins of the open water habitat of the lake) and the wetland and reedbed systems.

4.5.3 Conceptual model of site ecohydrology

Analyses of the relationships between the contemporary ecology and hydrology across the wetland habitats have revealed the water regime as the key factor influencing the emergent macrophyte communities at Sheskinmore Lough. In contrast, water depth and exposure frequency are the primary factors impacting the submerged macrophyte communities within the lake habitat. Synthesis of the ecological and hydrological characteristics and relationships at Sheskinmore within a conceptual model of ecohydrology permits a broader assessment of the site (Figure 4.55). Based on spring-summer conditions (March to September), which represent the active season of vegetation growth and the period when wetland plants are most sensitive to the oxygen status in their root zone and therefore to waterlogging (Gowling et al, 2012), the lake-wetland system is divided into three core zones: 1) the Terminal Wetland - the western end of the wetland that is removed from significant direct lake hydrological influence; 2) the Transitional Wetland - the central wetland area including the reedbed habitat where the lake hydrology appears to have an important impact; and 3) the Lake open water system. Each zone is connected via surface and groundwater flows; however the connection between the terminal wetland zone and the lake is via the transitional wetland zone. The influence of the sluice is considered within the catchment processes for each zone to demonstrate its relative impact.

During the summer, there is an increase in ecological productivity in both the lake and wetland at Sheskinmore Lough. The nature of that increase in the transitional wetland and lake, however, differs depending on whether the sluice is open or closed. For example, significant summertime precipitation in the transitional wetland results in surface water inputs leading to soil waterlogging; however when the sluice is open, the levels of waterlogging are significantly reduced. The effect is an increase in *Fen meadow* and *Sedge mire* communities, which favour moist soil conditions, instead of an increase in *Peat mire* and *Rush mire* communities that favour wetter soil conditions. The model also highlights the effect of disturbance during these sluice induced hydrological fluctuations. In particular, the invasion of *Phragmites australis* in the transitional wetland, at the periphery of the reedbed, is enhanced by disturbance created from the large shifts in water level (Minchinton & Bertness, 2003). Disturbance-induced invasion is especially prevalent during periods when the sluice has been closed for sufficient time to maintain water levels at a depth that exceeds the tolerance range of emergent macrophytes. When the water levels finally recede, *Phragmites australis* is then well placed, due to its fast growing competitive nature, to invade the recently flooded areas (Whyte et al, 2008). In the terminal wetland, where groundwater dominates the ecohydrology, wet summer conditions result in the addition of surface water inputs as well as groundwater inputs. This leads to an increase in species and communities (*Peat mire*, *Rush mire*) favouring wetter soil conditions.

Similarly, lake ecohydrology is dependent on water management at the sluice. During wet summers, surface water inputs significantly raise the water level of the lake. The maximum level reached is much lower when the sluice is open, and the impact on ecology (increased *Littoral lowland* and *Coastal lowland* communities) is similar to when the sluice is closed during dry summer conditions (increased *Benthic coastal* and *Benthic lowland* communities), as both promote communities that prefer shallow to moderate depths. When the sluice is closed during a wet summer, water levels can rise as high as reached in winter. The effect is an increase in the communities favouring deep water environments, which has the potential to include the rare *Najas flexilis* (Wingfield et al, 2004). The *Najas flexilis* population within the lake could increase during dry summers, but only when the sluice is closed and water levels are maintained at moderate depths. In the dry summer months the transitional wetland is also impacted by the lower than average water levels in the lake, with soil drying resulting in increases in communities which thrive on groundwater inputs (e.g. *Fen meadow* and *Groundwater mire*), and this is especially likely when the sluice is open.

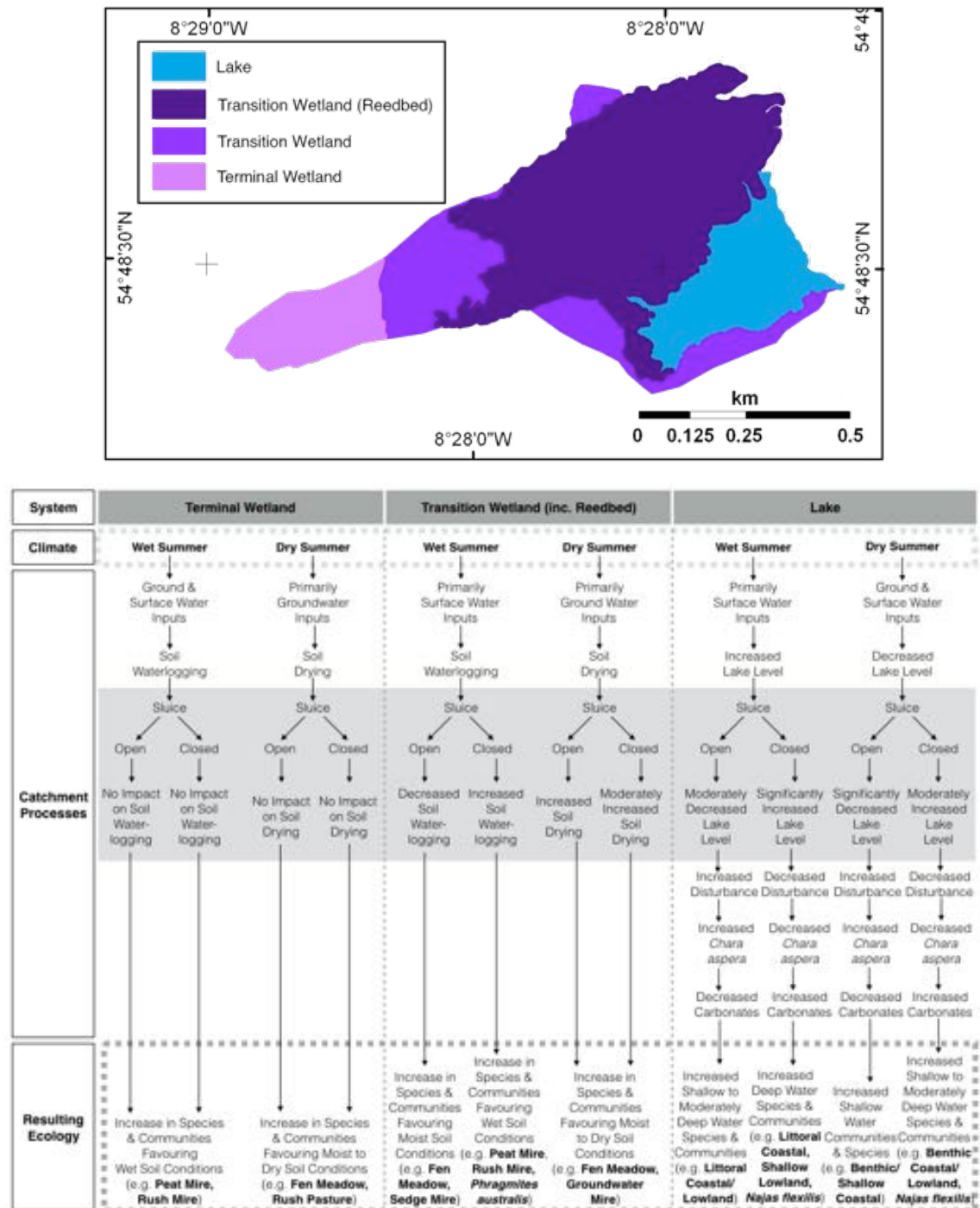


Figure 4.55 Conceptual ecohydrological model highlighting the key relationships between the ecology and hydrology of the wetland and lake system. The accompanying map shows the distribution of the system zones in the model, which include the terminal wetland, transition wetland (which includes the reedbed) and lake.

4.5 Conclusions

The results in this chapter have revealed a lake and wetland system that has a complex contemporary ecohydrology set in a complicated coastal environment. The large array of coastal and terrestrial habitats, from extensive aeolian-derived sand dunes to hilly peatland, provides a plethora of niches within which a multitude of vegetative species and communities survive. The lake and wetland system overlays the geological transition from

calcareous dunes to acidic peat, which overlay siliceous bedrock. In the wetland, the species and communities identified indicate fen and mire communities typical of oligotrophic hydrologies with circumneutral pH levels and low conductivities, and peaty or sandy calcareous substrates. Analyses of pH, conductivity and nutrient data collected in situ confirms these findings. They also confirmed the ecologies of the communities identified in the lake, as these too are indicative of oligotrophic, circumneutral pH levels and low conductivities on peaty or sandy substrates. Classification also revealed the hydrological requirements of the communities identified across the lake and wetland system. In the wetland, hydrological inputs and topographical variation largely determine the distribution of the communities. The highest diversities were found at the western, or terminal, end of the wetland where groundwater influence is significant and the lowest diversity found at the lower elevations in the centre of the wetland nearer the lake where groundwater flow is primarily influenced by the lake. Disturbance from grazing and land management at the western end of the wetland may also play a part in enhancing community diversity in that area. Within the lake, the highest community diversities are found in the areas of moderate depth, especially on the slopes within the channel feature and in the areas near the river inlets. Littoral communities, including the endangered *Najas flexilis*, thrive on the organic silt substrates at the base of the channel, while charophytes dominate the sandy substrates in the shallow zone where diversity is consequently extremely low.

Most notably, this study has identified that the operation of the sluice causes water levels to fluctuate by up to 1m, with rapid declines in water level observed once the sluice has been opened. The knock on effects of these significant lake level fluctuations have been observed at hydrological monitoring sites surrounding the lake, as well as at those located a much further distance west of the lake and reedbed within the central wetland area. Only the terminal western end of the wetland and surrounding areas appear to be unaffected. In addition, the impacts from this hydrological disturbance are reflected in the site ecology especially the distribution of macrophyte communities, the abundance of *Chara aspera* in the lake, the endangered status of *Najas flexilis*, and the encroachment of *Phragmites australis* across the wetland.

5 Multi-proxy paleolimnology of Sheskinmore Lough

5.1 Introduction

The environmental history of the lake and wetland system was assessed using paleolimnological techniques to achieve an improved understanding of the evolution of this freshwater system from what was understood to be estuarine conditions over 1000 years ago (Shaw & Carter, 1994). This chapter describes and attempts to explain the spatial (intra-/inter-site) and temporal (downcore) variations and trends in the sedimentary sequences of the lake and wetland system. Stratigraphies are presented for eleven exploratory sediment cores obtained from the lake and wetland system in June 2012 to gauge the preservation potential of selected paleoecological indicators. Stratigraphies are also presented for two deeper cores extracted from the deepest part of the Sheskinmore Lough in June 2013. This is followed by more intensive, sample-based analysis, culminating in chronological evaluation of one of the lake cores. The ultimate goal of this chapter is to gain a better understanding of historical variability in the environmental and ecology of the lake, and in particular to capture these for the decades preceding installation of the sluice and implementation of water management.

5.2 Wetland and lake sediments and stratigraphy

Spatial (intra-/inter-site) and temporal (down core) variations and trends in sedimentary sequences across the Sheskinmore wetland system were examined through the analysis of sediments and paleoecological indicators in eleven exploratory sediment cores (Figure 5.1). These exploratory cores were retrieved in part to assess the feasibility of more intensive sample-based palaeoecological and chronological analysis, but also provide a more comprehensive, spatial context regarding the organisation of sedimentary environments and evolution of the system.



Figure 5.1 Locations of the exploratory sediment cores retrieved in June 2012 from hydrological monitoring sites across the wetland (S3, W1, W2, W3, W4, W6, W7) and within Sheskinmore (L3, L3.2, L3.6) and Sandfield (L6) Loughs. Note W1 and W4 did not undergo further analyses, only sediment description, due to contamination.

5.2.1 Sedimentology

Figure 5.2 shows downcore lithostratigraphies of the eleven exploratory sediment cores. Sedimentological descriptions were based on a modified Troels-Smith scheme described by Long et al (1999). Although the length of each core differs significantly, there is a clear distinction in sediment type between the lake and wetland cores. The 30 cm cores extracted from Sheskinmore Lough (L3.6, L3.2 and L3) and L6 extracted from Sandfield Lough are all dominated by organic sand. L3.6 and L3.2, located in the flat western side of the lake, also contain a surface layer of sand, with the latter overlain by a 0.1m layer of silt. In contrast, the top 0.3m of the wetland cores (W1-4 with the exception of W6) is dominated by organic peat sediment. The top section (0-0.4m depth) of each of those cores also contain sections of peaty sand and sandy peat. A fine band (0.05m) of pure sand is present between 20 cm and 0.4m depth in cores W3 and W4. In contrast, W6 located at the western side of the open water habitat of the lake, contains a top core stratigraphy dominated by organic sand that is more akin to those collected from the lake. It can, however, be distinguished from the aquatic lake cores due to the 0.05m layer of organic peat present at the surface.

From an elevation perspective, and with the exception of core L6, which is located at a much higher elevation (5.5-5.8mOD), the organic sand-dominated lake cores are all within the elevation range of 2.3-2.8mOD. The top 75cm of W6, which also contains a large proportion of organic sand, spans the top section of this lake core elevation range, between 2.4 and 3.2mOD. The bottom two thirds of W6 (0.75m to 2m) is characterised by alternating bands (0.05-0.2m) of organic peat, sandy peat and pure peat, underlain by organic sand. Similarly, W7 (which is at a similar surface elevation to W although the top section (0-1.3m) of the core is missing) is characterised by peat, peaty sand and sandy peat banded layers (0.05-0.2m) to 1.7m depth, and is also underlain by organic sand.

The cores located towards the western end of the wetland system (W1-4) contain a 0.2m surface layer of organic peat, except for W3, which has a thinner surface layer (0-0.1m) and underlain by sandy peat (0.1-0.3m). These wetland cores then comprise layers of peaty sand, sandy peat and silty peat of variable extents and at variable depths. W3 and W4 also contain a narrow (0.05m) band of sand in the upper layers, and W2 and W3 both terminate in sand. These relative similarities in sedimentary sequence occur despite notable differences in elevation between the cores, with >3m rise in between the centre and west of the wetland. Core S3, located to the northwest of the wetland represents the maximum elevation (8.3mOD) of the core sites, and differs from all the other cores as it comprises only sand.

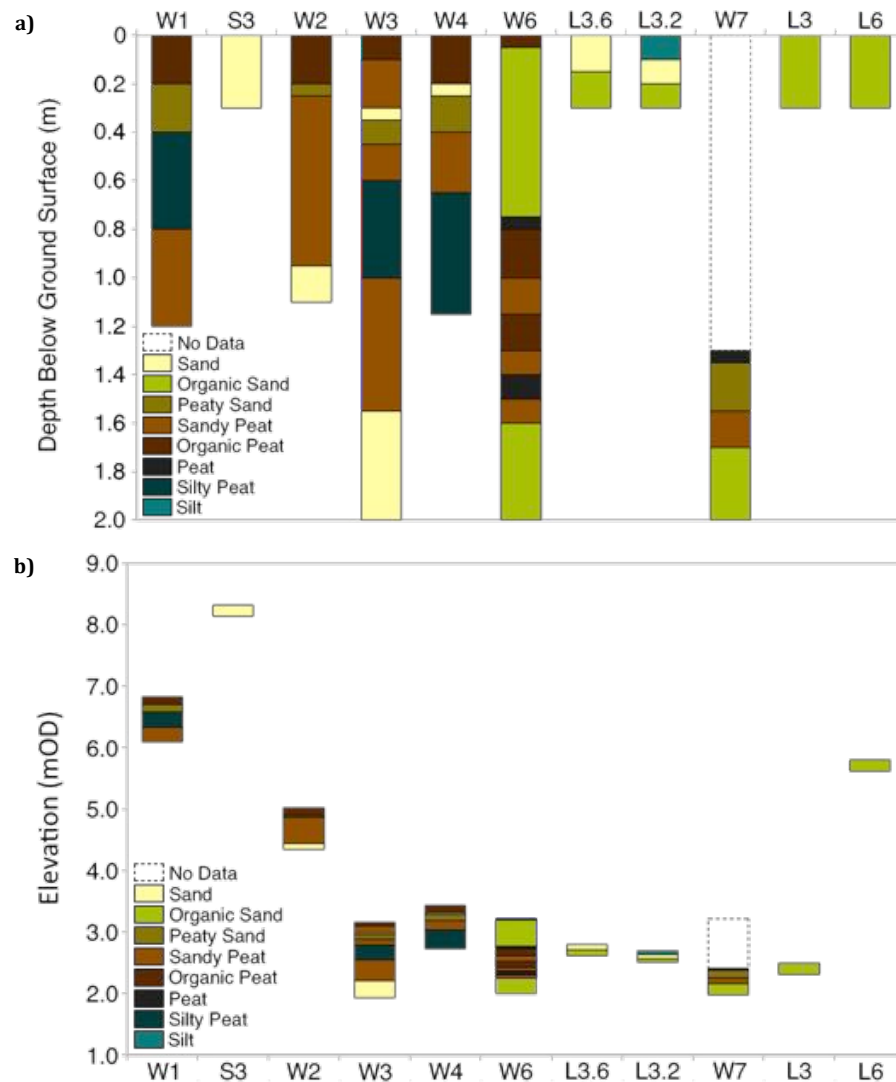


Figure 5.2 Stratigraphy of the exploratory sediment cores extracted from sites across the lake and wetland system relative to a) the ground surface and b) OD. Cores are ordered west to east (from left to right) with W1 being the most westerly site and L6 being the most easterly site. Note Appendix E. provides more detailed measurements of the sections.

Grain size distribution across the site is fairly uniform (Figure 5.3) and dominated by fine (>52%) and medium sand (>17%). Samples from the Sheskinmore Lough core L3 also contain a small amount of coarse sand (1%); this fraction is absent in all other cores. All samples contain low proportions of the finer sand and silt fractions. L3 contains the highest proportion of very fine sand (15%) and also contains small proportions of silt (1.3% coarse silt; 2.8% medium silt; 1.1% fine silt; 0.8% very fine silt) and clay (0.6%); L3 is thus poorly sorted. In contrast, the other samples within Sheskinmore Lough (L3.2 and L3.6) contain no silt or coarse sand fractions and are therefore better sorted. Sandfield Lough (L6) also comprises a small proportion of fine sediment (1.2% coarse silt; 2.5% medium silt).

Similar to cores L3.6 and L3.2 from Sheskinmore Lough, W2 and S3 are well sorted and contain no silt or clay fractions. Core W3, similarly dominated by sand fractions, comprises a fine layer close to the surface (0.1-0.3m depth) containing clay (0.2%) and silt (4.2%).

Cores W6 and W7 contain the largest proportion of fine material, the former exhibiting down-core fining and the latter up-core fining (although W7 only represents sediments at depth). Increases in the range of different size fractions are associated with a decrease in sorting.

Organic content of substrates across the site is relatively low with the majority of samples containing less than 20%. There are sites that exhibit very low organic content in surface layers (<5% at S3, W6, L3.2 and L3.6) and some containing more substantial organic fractions (10-20% at W2, W3 and L3). Within the stratigraphies, those layers containing notable fine material also comprise more substantial organic matter (e.g. 1-1.5m depth at W6 and W7). The low levels of surface organic matter may indicate recent deposition of windblown, organic-poor sand across the site. The high percentage (21.8%) of surface organic matter at the surface of core W3 however, indicates the distribution of aeolian sand deposition is not uniform across the site. The carbonate content shows little association with the organic content, but is very low (<1%) across most samples with little variation across the site. The lakes contain slightly different carbonate contents (0.9% at Sheskinmore (L3) and 2.3% at Sandfield (L6) Loughs). The only notable exceptions are layers at depth (1.2-1.8m depth) in W3 and W7, which contain 2.5-4% carbonate.

5.2.2 Paleoecological indicators

An advantage of using diatoms as a proxy for reconstructing environmental change is that they tend to preserve well in sediments and, as a result, are a valuable paleoenvironmental indicator (Vos and de Wolf, 1993). Battarbee et al (2001), however, state that in extreme cases, some diatom species are completely absent from the sediment record due to dissolution problems. The accuracy with which diatom assemblages in lake sediments represent the composition of the source communities from which they were derived is therefore a potential issue, especially as preservation in lake sediments is largely determined by lake pH. Conversely, the degree of preservation can act as an indicator of the paleoenvironment itself, especially in the coastal environment where one of the main problems is distinguishing between autochthonous and transported allochthonous diatom species (Freund et al, 2004). For instance, Freund et al (2004) used the percentage of broken diatom frustles as an indication of the energy level of the environment and subsequent transportation, whilst also emphasising the importance of fragile species as indicators of the paleoenvironment given the species once lived at the place of deposition.

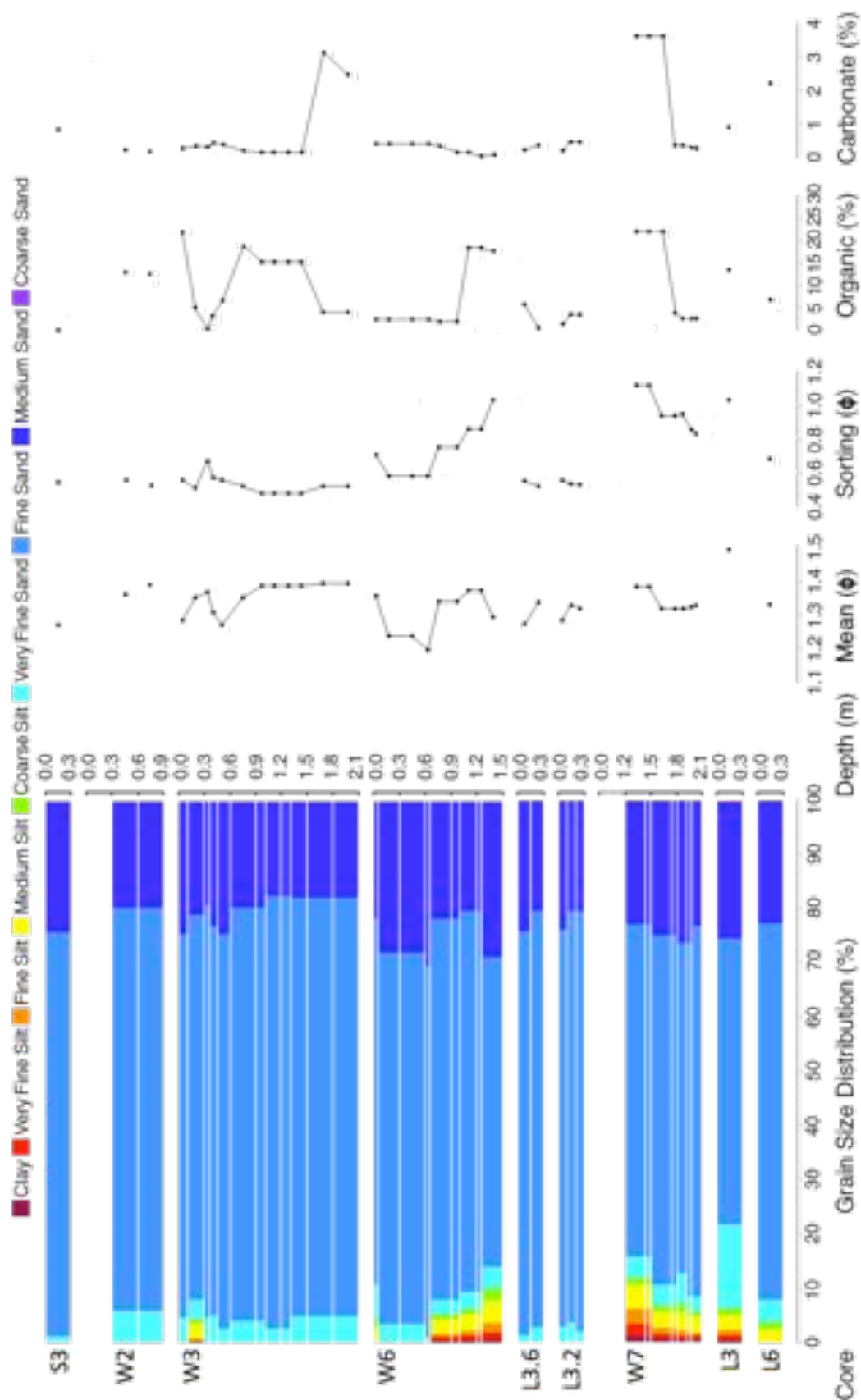


Figure 5.3 Exploratory core grain size distribution, mean grain size, sorting, percent organic and percent carbonate content in relation to depth below the ground surface. Note, data are presented vertically to aid visualisation; samples within the individual cores are vertically related, but there is no vertical association between cores.

The macrofossils identified in the exploratory cores reveal that the preservation potential of these proxies as paleoecological indicators in this study is high within the lake sediments. Like diatoms, the macrofossils have preserved best within the open water lake habitats, with few identifiable specimens found within the exploratory cores across the wetland habitat. The macrofossils identified within sample L3 (0-30 cm depth) reflect the current ecological lake conditions established in the contemporary ecological survey (Figure 5.4). The large abundance of insect fragments (65 individual pieces) and Trichoptera fragments (26 individual pieces) indicate a shallow lake ecosystem dominated by species favouring water depths of just a few metres (Wiberg-Larsen et al, 2001).

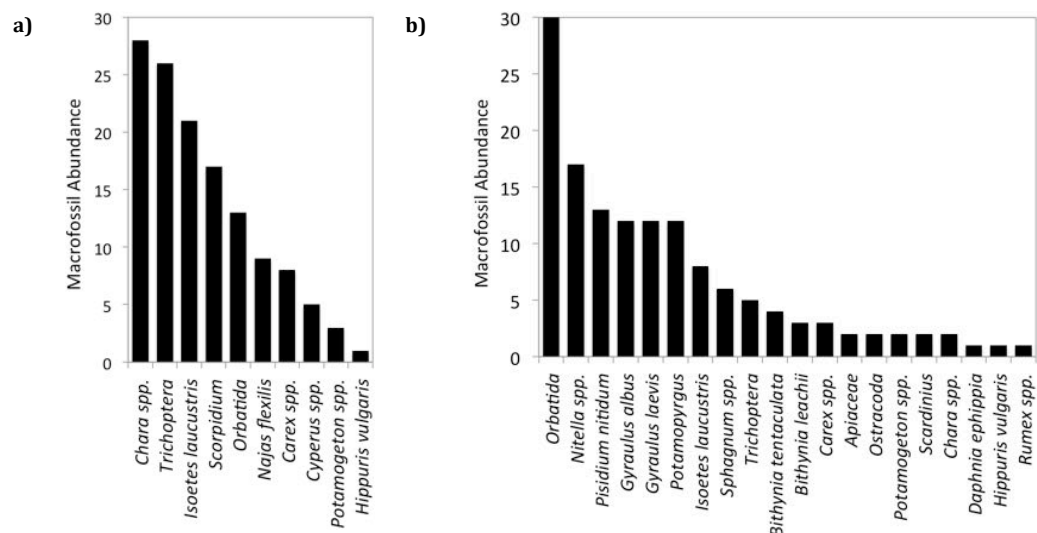


Figure 5.4 Macrofossil abundance identified within the top 30 cm of sediment taken from a) Sheskinmore Lough (L3) and b) Sandfield Lough (L6).

The abundance of *Chara* oospores (28 individuals) and *Isoetes lacustris* megaspores (21 individuals) reflects the present shallow lake waters, macrophyte dominance by *Chara aspera* within the lake and the influence of the surrounding calcareous environment of the sand dunes (Huang, 2002; Weckström et al, 2010). The presence of *Potamogeton* spp. (3 seeds) reflects the present oligotrophic freshwater conditions observed within Sheskinmore Lough. Although scarce, they are nonetheless significant as their scarcity within sediment records is not uncommon due to lower seed production and limited dispersal compared to other macrophytes (Weckström et al, 2010). Similarly, the presence of *Najas flexilis* seeds (9 seeds) corroborates the macrophyte survey findings and is important as the dispersal of *Najas flexilis* seeds is limited (Wingfield et al, 2004). The macrofossils identified within the Sandfield Lough sample L6 (0-30 cm depth) also reflect the contemporary ecological lake conditions already noted. The abundance of *Chara* oospores (240 individuals), which includes 83.3% that are calcified, emphasises the considerable calcareous influence within Sandfield Lough due to its close proximity to the estuary and sand dune system. The higher diversity of macrofossils in Sandfield Lough, compared to Sheskinmore Lough, is likely due to the slightly more eutrophic conditions observed there, as evidenced in the diatom species assemblages.

In accordance with the findings of the sediment stratigraphical analyses, the abundance of paleoecological indicators within the exploratory lake cores, especially L3 and L6, also confirms that more intensive sample-based stratigraphic analysis is possible. In particular, the high preservation potential of diatom and macrofossil indicators within the lake sediments shows that, via detailed analysis of complete cores, this study will be able to build a more complete picture of the site environment in the past, and also its ecohydrology.

The diatoms identified in the exploratory cores also reveal that the preservation potential of diatoms as a paleoecological indicator in this study is high within the open water lake habitat. Their preservation potential, however, is extremely poor across the wetland habitat, so much so that only a few diatom fragments were identified in the majority of the wetland cores. The diatom species identified within the Sheskinmore Lough, specifically at exploratory site L3 (0-0.3m depth) and at the Sandfield Lough exploratory site L6 (0-0.3m depth), are indicative of freshwater conditions (Figure 5.5). Sheskinmore Lough is dominated by macrophyte-dwelling species, especially *Fragilaria exigua* (15%), *Tabellaria flocculosa* (12%) and *Navicula radiosa* (10%). Other frequent species include *Diatoma vulgare* (9%), *Cymbella microcephala* (6%), *Fragilaria capucina* (5%), *Achnanthes helvetica* (5%), *Eunotia implicata* (5%) and *Fragilaria brevistriata* (4%). These species indicate an oligotrophic environment with a pH of between 6 and 8 (Knudson & Kipling, 1957; Flower et al, 1996; Barbiero, 2000). In contrast, Sandfield Lough is dominated by *Achnanthes lanceolata* (15%), *Cocconeis placentula* (13%), *Diatoma vulgare* (10%), *Achnanthes minutissima* (9%), *Navicula cryptocephala* (9%) and *Fragilaria construens* (9%). These species indicate an oligotrophic to slightly mesotrophic aquatic environment with a pH of between 7 and 8 (Patrick, 1968; Moore, 1977; Kelly, 1995).

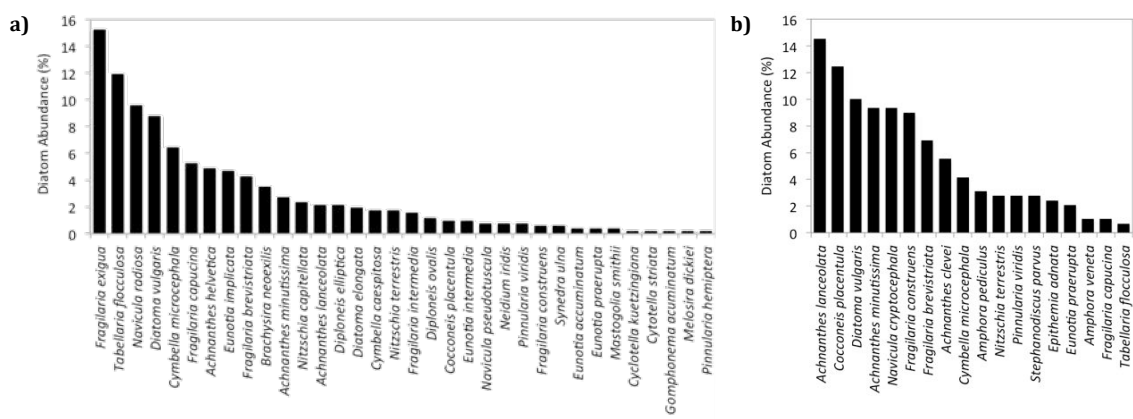


Figure 5.5 Diatom species abundance identified within the top 30cm of sediment taken from a) Sheskinmore Lough at L3 and b) Sandfield Lough at L6.

Like diatoms, macrofossils also have several advantages as paleoecological indicators for reconstructing past environments. Seeds and fruits can often be identified to genus or species level, whilst many common pollen types can usually only be identified to family or genus level (Birks, 2001). In addition, flora such as bryophytes and Characeae that are not

represented by pollen or spores produce macrofossils that are important constituents of vegetation. Macrofossils are particularly useful to this study as they are less readily dispersed than most types of pollen and so they tend to represent the local vegetation within a lake and its surrounding catchment. This provides a more precise definition of the past local vegetation (Birks, 1973; Birks & Birks, 1980). Macrofossils therefore extend the floristic detail and the ecological value of palaeolimnological reconstructions (Jones et al, 2007). Like diatoms, the degree of preservation can act as an indicator of the paleoenvironment itself. Vernimmen (2002) looked at the quality of macrofossil samples and identified factors that affected deterioration. They proposed five categories ranging from very poor (the sample contains fragments only and no recognisable seeds) to very good (the seed surface structure is still entirely intact). Although this system seems fairly simplistic, it allowed comparisons between samples. For example, in most cases there appeared to be a positive correlation between water table and preservation of botanical remains, and samples with higher proportions of organic matter generally appeared to have better preserved macrofossils (Vernimmen, 2002).

5.2.3 Summary

Analyses of the exploratory cores revealed that more intensive sample-based stratigraphic and palaeoecological analyses are feasible. Down core variations in sedimentology reveal distinctive layers that vary not only in situ with depth, but also vary spatially. Furthermore, the large abundance and variety of diatom species and floral and faunal macrofossils observed within the exploratory cores from Sheskinmore and Sandfield Loughs supports further paleoecological analyses in this study.

5.3 Lake sediment stratigraphy

5.3.1 Lithostratigraphy

As a result of the exploratory study, two lake cores were extracted in June 2013 from the deepest parts of Sheskinmore Lough, specifically from the base of the channel feature in the northern and central southwestern sections of the lake. This was deemed an appropriate number as other studies have shown that past lake vegetation can be suitably represented by two cores (Davidson et al, 2010; Sayer et al, 2010). In addition, submerged vegetation within the lake exhibits low spatial heterogeneity, therefore two cores could adequately reconstruct past changes at the scale of whole lake (Sayer et al, 2010). Coring locations were chosen on the basis that one was taken from a deep and one from a shallow part of the lake channel feature in order to capture both lake-wide changes in macrofossil composition and diatom assemblages, as the latter especially, are spatially less variable in small lakes (Anderson, 1986; Sayer et al, 2010). Appendix F. contains detailed stratigraphic information for the two complete lake cores including geographic position and extraction particulars.

Cores were split in half vertically and photographed before description, assessment and analysis (Figure 5.6). Sedimentological descriptions of the texture and composition of each unit were based upon a modified Troels-Smith scheme described by Long et al (1999). Light and dark layering is clearly visible in the photographs, which is evidence of shifts from sand- to organic-rich beds in the stratigraphy. Core 1 is the deeper core of the two cores, both in terms of its surface elevation (at 2.62mOD, this is one of the deepest parts of the lake) and its extent (a core length of 65cm brings the base of the core to 1.97 mOD). The surface sediment here is dominated by a thin layer of fine organic silt that is underlain by a thicker later of organic sand. Core 2, which was obtained at a lake bed elevation of 2.90mOD within the southwest portion of the channel feature, extends 46.5cm to a depth at 2.43mOD, and also has a thin organic silt surface layer underlain by a more modest unit of organic sand.

Below the lakebed surface, there is a complex transition over 10-20cm into pure peat. In Core 1 this transition is fairly rapid, with multiple thin layers (1-1.5cm) progressing from peaty sand, to pure sand to sandy peat. In contrast, this transition is more gradual in Core 2, with peaty sand progressing to pure sand and then sandy peat via layers that are, in some cases, twice the thickness (1-3cm) as those in Core 1. Also, there are more stages to this transition, with additional layers of sandy peat and organic sand, as well as peaty sand layers bordering the peaty sand layer, which lies just above the peat layer. Apart from a thin later of peaty sand at 26-28cm depth, the peat layer in Core 1 is uninterrupted from between 16cm depth and 57cm depth. In contrast, the peat unit in Core 2 only spans 21-29cm depth. Similar to Core 1, however, the peat layer in Core 2 is overlain by a thin layer of peaty sand (18-21cm depth), which itself is topped by sandy peat. Below the peat layer the cores differ again as the peat section in Core 1 is underlain by sandy peat, progressing to a thin layer of pure sand (1.5cm) and then transitioning back to peat at the base. The peat layer in Core 2 is also underlain by sandy peat, but it then progresses gradually to a large unit of pure sand (8.5cm) at the base via thick layers (4cm) of peaty and organic sand.

The lake sediment cores are dominated by sand-sized material. The top 29cm of Core 1 is dominated by fine sand (<53%), medium sand (<35%) and very fine sand (<29%) with almost no clay fraction (except for a trace at 27cm to 28cm depth), and very small coarse sand silt fractions (Figure 5.7). The mean grain size in this section is consistent at around 1.3Φ (0.4mm) and sorting is moderate (0.87Φ); however halfway down the section, from 14cm depth downwards, there is an increase in the coarse silt and very fine silt fractions. Sediment here is similar to the top 2cm of the core, and the surface also contains the largest proportion (3%) of coarse sand in the top 29cm. In the middle section (29cm to 48cm depth) the smaller silt-sized sediment fractions increase, encompassing up to 30% of each sample. Mean grain size fluctuates from 1.1Φ (0.5mm) to 1.7Φ (1.3mm) in this section, and sorting is generally poor (1.1Φ). Some layers (29-30cm, 36-37cm, and 38-39cm depth) include a clay fraction, albeit in small proportions (<2%). This middle section is also characterised by an increase in coarse sand of up to 7%. The coarse sand sediment fraction continues downwards to 61cm depth where it is absent from the deepest section of

the lake, except for between 63.5cm and 65cm where a tiny proportion (1%) is present. From 48cm depth to 52cm depth there is a decrease in silt and this is reflected in a slightly larger mean grain size (1.0Φ ; 0.5mm) and poor sorting (1.3Φ). In contrast, the next 11.5cm of the core is characterised by a gradual increase in sand, with the last 1.5cm displaying silt proportions akin to some of the samples within the section between 29cm and 48cm depth. Here there is virtually no silt, however the proportions of coarse sand are also very small, averaging at around 3%. Instead it is the fine sand that dominates at around 60% along with medium sand at about 35%.

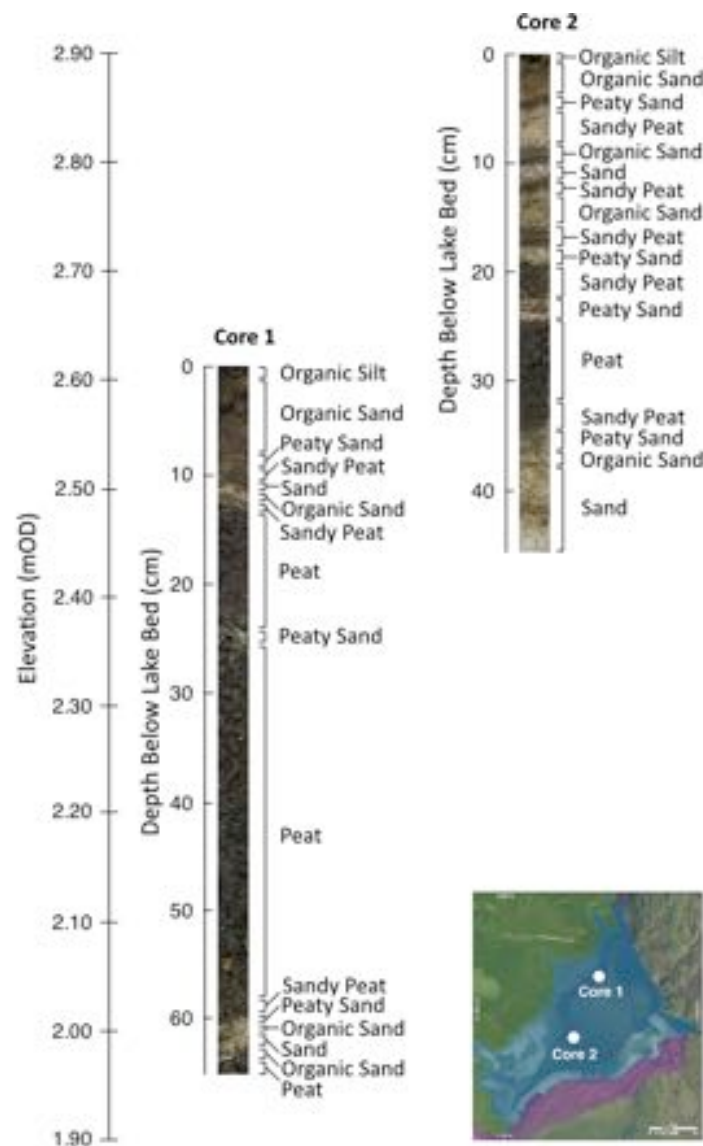


Figure 5.6 Sediment stratigraphies of the two Sheskinmore Lough lake cores with elevation. Images show the split core surface.

In contrast to the significant variation in Core 1, the downcore sediment size of Core 2 is much more consistent (Figure 5.8). Apart from the surface sample, which contains larger proportions of silt and very fine sand, the remaining top 21cm of the core is dominated by fine sand (up to 50%). This is reflected in a consistent mean grain size of around 1.2Φ

(0.4mm) and moderate sorting around 0.7Φ . Silt is completely absent from 13.5-14cm depth and 19.5-20.5cm depth where medium sand increases from ~30% to ~40%. There are layers within depth ranges 21-24cm and 26-30cm where finer sediment fractions are evident, with silt forming up to 20% of the sample. Although the proportion of fine sand decreases in these sections, the proportion of medium sand does not change. From a maximum fines component at 27-28cm, the very fine sand and silts phase out with depth and are absent below 36cm. The lower section of the core is almost entirely formed of fine and medium sand, except at the base (45-46.5cm depth) where coarse sand increases from 4% to 18%.

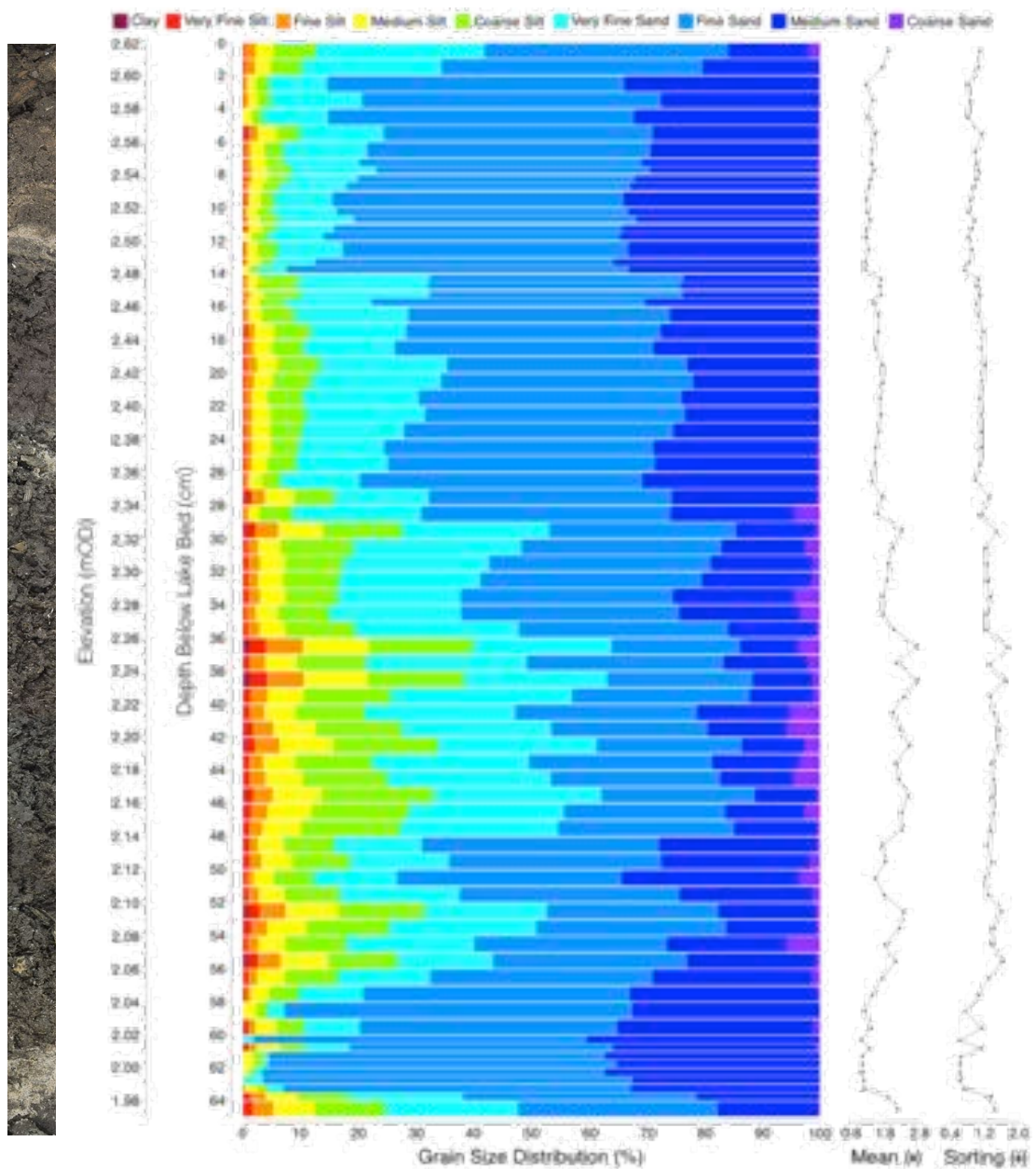


Figure 5.7 Lake Core 1 grain size distribution, mean and sorting relative to elevation (mOD) and depth below the lakebed (cm).

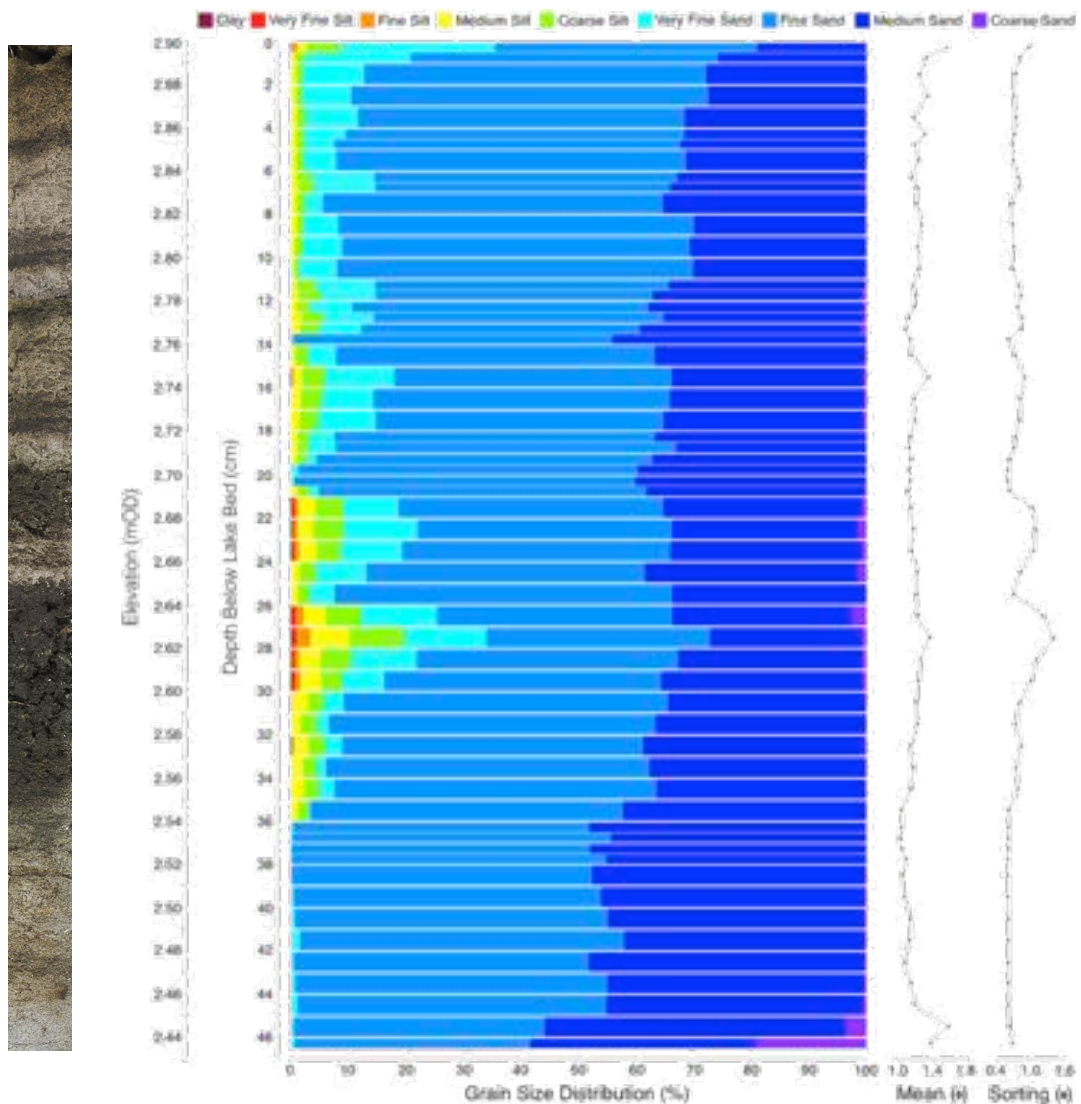


Figure 5.8 Lake Core 2 grain size distribution, mean and sorting relative to elevation (mOD) and depth below the lakebed (cm).

Organic and carbonate content were analysed for the two lake cores which showed that on average, Core 1 contained more organic material than Core 2 (Figure 5.9). Organic content within the top 20 cm of Core 1 is low, averaging at 9% with a small peak at the top (14%). Similarly, Core 2 comprises low organic content, averaging at just 2% between the surface and 23cm depth, again with a peak occurring in organics at the surface. In Core 1 there is a gradual and variable increase in organic content from 20cm to a peak of 55% at 40cm depth. A sub-surface peak is also evident in Core 2, but marks a sharper increase in organic matter from low at 23cm to a maximum of 36% at 28cm depth. The decrease in organic matter after this point is equally rapid as by 33cm depth it reduces to <5%, declining further to <1% at the base of the core. There is also a decrease in organic matter from the mid-core peak in Core 1, but the decline is complicated by continued fluctuations around a declining mean of up to 10%. A minima of 3% is reached at 63cm depth after which organic matter increases to 35% at the base of the core.

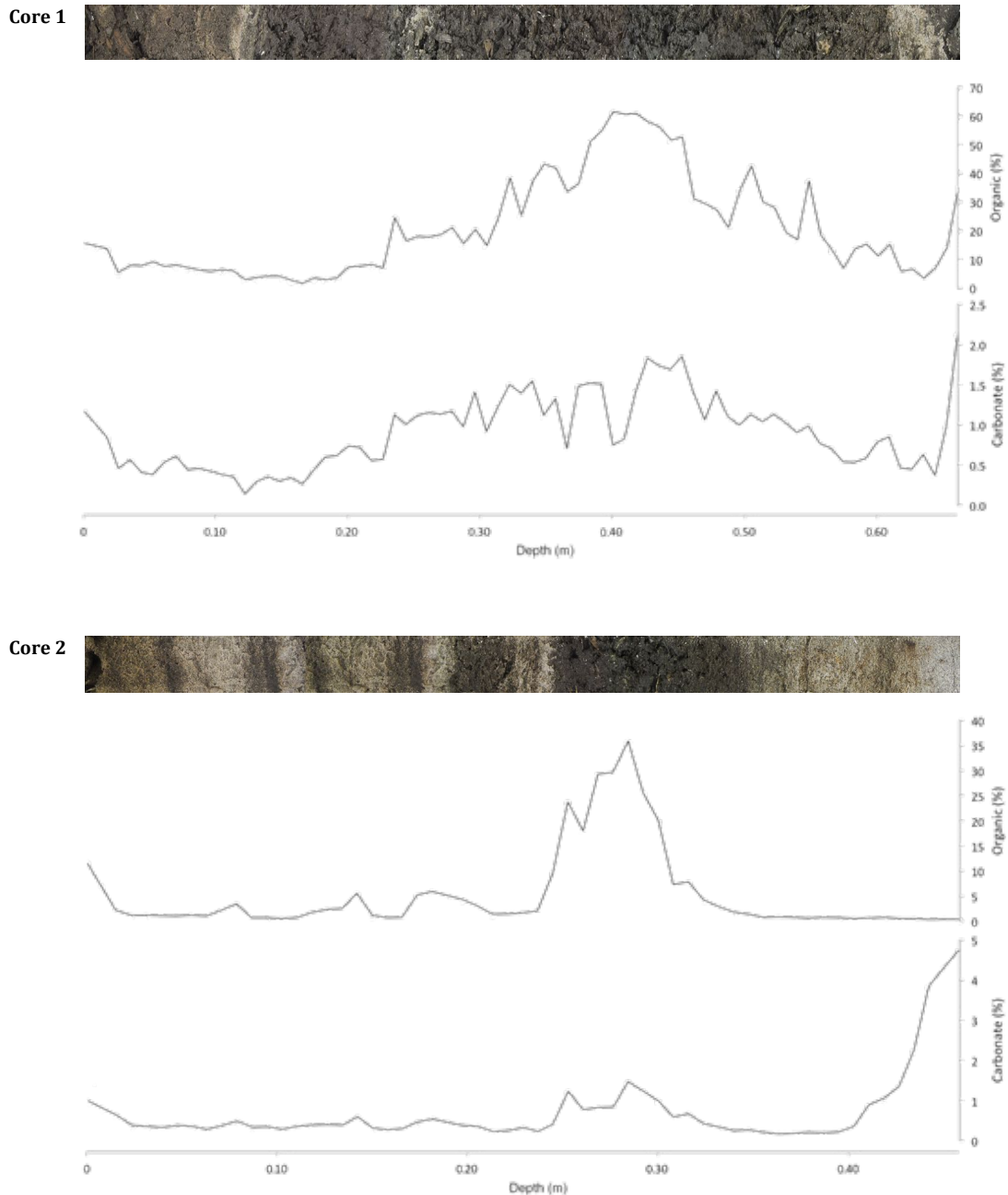


Figure 5.9 The percent organic and percent carbonate content in Core 1 and Core 2 in relation to depth (m) below the lake bed. Note the different y-axis scales.

The downcore pattern of carbonates in Core 1 is very similar to the organic matter content and mirrors the overall trend with one or two notable differences. First, the levels of carbonate found throughout the core are very low and do not exceed 2.5%. Second, a peak in carbonate content between 40cm and 50cm depth is less pronounced than the mid-core peak in organics, and is interrupted by multiple dips in carbonate levels. Third, the decline in carbonate levels to the base of the core is more consistent, with a peak of 2% in the last sample. The carbonate content in Core 2 also largely mirrors the downcore pattern in organic matter with a low baseline (<1%) and mid-core peak to 2% at 23-32cm depth. At the base of the core however, there is a relatively rapid rise in carbonate content to a maxima of 4.75%.

5.3.2 Palaeoecology

5.3.2.1 Macrofossils

Macrofossil counts were made from the lake core samples. In the 76 samples comprising Core 1, 50 macrofossil types were identified and in the 58 samples comprising Core 2, 31 macrofossil types were identified. In total, this equates to 59 unique macrofossil types. Figure 5.10 illustrates the degree of overlap between the two cores, and also distinguishes between animal-based and plant-based macrofossils, and aquatic, marginal or terrestrial (Haslam et al (1975), Phillips (1980), Jeremy et al (1982), Fitter et al (1984) and Rose (2006)). Core 1, which has a greater abundance and variety of macrofossils compared to Core 2, contains 15 aquatic (of which 6 are animal-based), 19 marginal, and 16 terrestrial macrofossils. In contrast, Core 2 contains 17 aquatic macrofossils (11 of which are animal-based), but only 10 marginal and 5 terrestrial macrofossils. Aquatic animal macrofossils exclusive to Core 2 include: the freshwater snail (aquatic pulmonate gastropod mollusc), *Bithynia leachii*; the class of aquatic (freshwater and marine) amoeboid protists, Foraminifera; the freshwater snail (aquatic pulmonate gastropod mollusc), *Gyraulus laevis*; the class of (freshwater and marine) Crustacea, Ostracoda; the freshwater clam (bivalve mollusc), *Iridium nitidum*; and an assortment of unidentified shell fragments. In Core 1, the only exclusive animal is the zooplanktonic water flea, *Daphnia ephippia* ('*Daphnia*' the genus of small planktonic crustaceans in the order Cladocera and '*ephippia*' the thick shell that encloses their eggs). Present in both cores is the freshwater bryozoan, *Cristatella mucedo*, the freshwater cyanobacteria, *Gloeotrichia spp.*, the aquatic insect class, Insecta, the aquatic beetle mite, Oribatida; and the order of insects, Trichoptera. It should be noted that Insecta and Oribatida could also be classed as marginal and terrestrial but given they were identified within lake sediment, they have been characterised here as primarily aquatic.

In terms of abundance, the macrofossils identified in Core 1 are dominated by *Phragmites australis* (31%), *Gloeotrichia spp.* (16%), *Scorpidium scorpioides* (12%) and *Nitella spp.* (11%) (Figure 5.11). Similarly, *Phragmites australis* dominates Core 2 at 50%, along with *Scorpidium scorpioides* (11%), Insecta (9%), *Chara spp.* (8%) and *Nitella spp.* (8%). *Gloeotrichia spp.* is less prevalent in Core 2 at just 7% and in Core 1, *Chara spp.* is less abundant at 6%. Oribatida is present in both cores, at 7% in Core 1 and 4% in Core 2, as is Trichoptera at 2% in both Core 1 and Core 2. The abundant macrofossils that are exclusive to Core 1 include *Daphnia ephippia* (3%) and *Juncus bulbosus* (2%), and exclusive to Core 2 include *Equisetum fluviatile* (1%) and Foraminifera (1%).

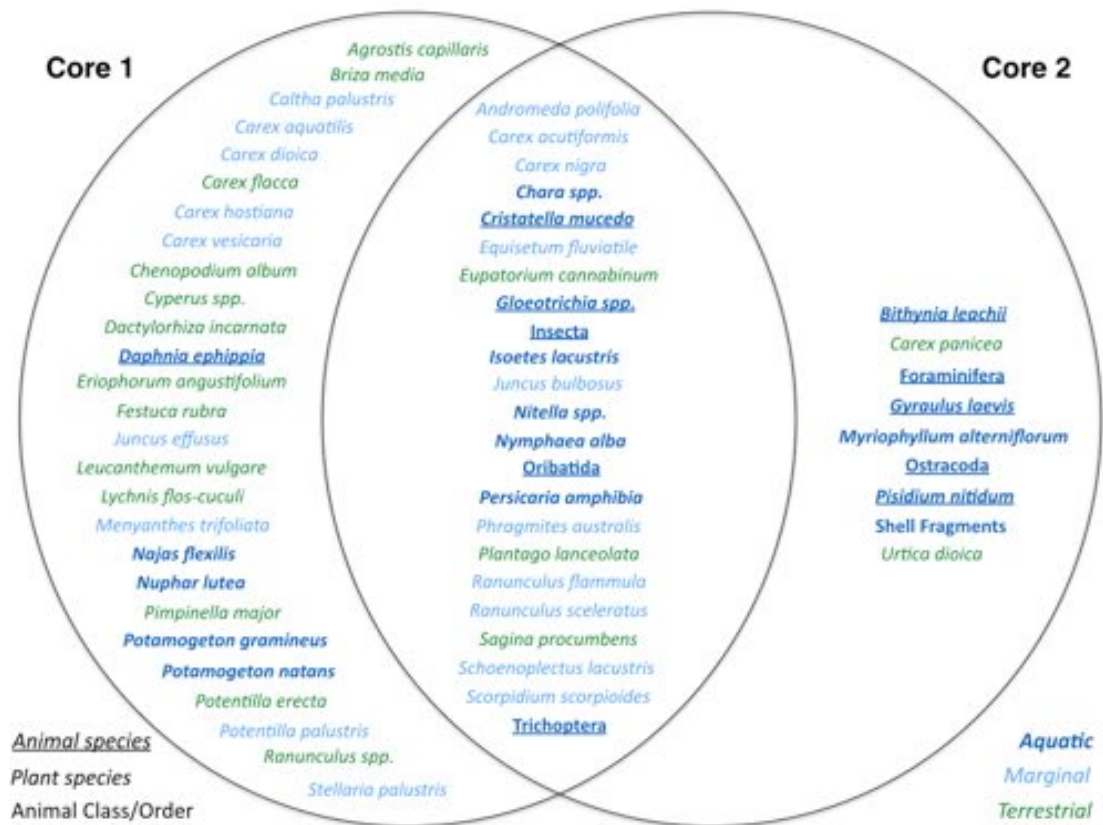


Figure 5.10 Venn diagram showing the 59 aquatic, marginal and terrestrial plant species and animal species/class/order of macrofossils that were identified only in Core 1 (27), only in Core 2 (9), and in both Core 1 and Core 2 (23).

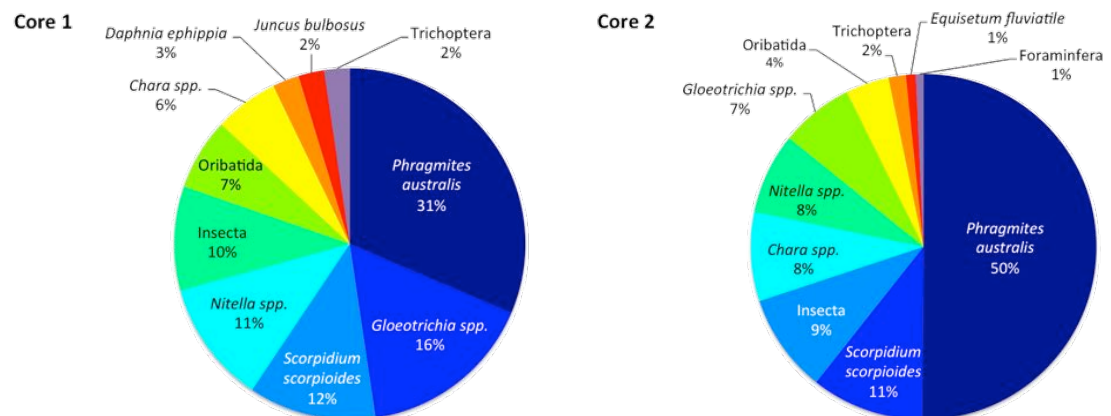


Figure 5.11 Pie charts showing the proportions of the 10 most abundant macrofossils identified in each of the two lake cores extracted in June 2013.

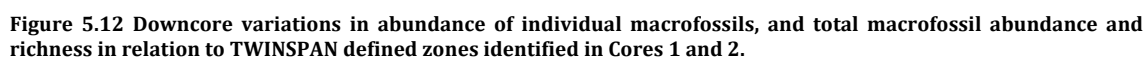
The total number of taxa observed in each core sample (per cm³) were plotted as stratigraphic diagrams for both cores using C2 1.7.4 software (Juggins, 2003). TWINSpan cluster analysis was performed on the macrofossil data combined from both cores to define groups of macrofossils and down core changes (Van der Putten et al, 2004; Hill and Šmilauer, 2005; Cañellas-Boltà et al, 2012; Ronkainen et al, 2015). The TWINSpan groupings are marked on the stratigraphic diagrams in Figure 5.12, which display the

relative downcore macrofossil abundance for both lake cores. The abundance of each macrofossil type in each sample was rescaled from 0 to 1 for the analysis, by setting the macrofossil with the highest abundance in each sample to 1 and calculating other individual macrofossil abundance values as a percentage of the highest abundance (Cañellas-Boltà et al, 2012; Ronkainen et al, 2014). Three key macrofossil groups were identified in each core, delineated as zones on the stratigraphic diagrams in Figure 5.12.

Zone 1 is dominated by *Chara spp.*, *Gloeotrichia spp.*, Insecta, *Nitella spp.*, Oribatida and *Scorpidium scorpioides*. Moderately abundant macrofossils include *Gloeotrichia spp.*, which increases in abundance with depth from the surface, *Isoetes lacustris* and *Juncus bulbosus*, which are present in similarly low numbers throughout the zone, both disappearing at frequent intervals. *Phragmites australis* is present in this zone, but in extremely small numbers. This zone represents the top 28cm of Core 1 and the top 24cm in Core 2. In Core 1 it is characterised by a moderate total macrofossil abundance and moderate to high richness in comparison to the other zones. In Core 2, macrofossil richness is comparatively lower. Also important in this zone in both cores is Trichoptera, which displays a regular downcore pattern of presence and absence, peaking between 7cm and 8cm depth in Core 1 and at 18cm depth in Core 2. *Daphnia ephippia* is sparse in Core 1, peaking between 19cm and 22cm depth, and is absent from Core 2.

In contrast to Zone 1, Zone 2 is dominated by *Phragmites australis*, which increases in abundance with depth. This zone represents the section at 28-59cm depth in Core 1 and the section at 25-40cm depth in Core 2. *Chara spp.* is fairly consistent throughout this zone in both cores but peaks at 9cm and 11cm depth and is absent at 19cm in Core 1. In Core 2 it peaks at 4cm and 18cm depth and is absent at 15cm. *Daphnia ephippia* increases in abundance from 36cm to 56cm depth in Core 1, as does *Scorpidium scorpioides*, which increases significantly in the same core, peaking at 45cm depth. Total macrofossil abundance in Zone 2 is the highest for both cores, with over 50 individual specimens identified in all samples in Core 1, peaking at 286 individual macrofossils identified at 55cm depth. In Core 2, *Gloeotrichia spp.* peaks significantly at 33 cm depth in Core 1, but is largely absent from Zone 2 in Core 2. This peak in Core 1 coincides with a significant increase in *Phragmites australis*, which dominates this zone in both cores, increasing in abundance with depth and peaking at 55cm depth in Core 1 and at 33cm depth in Core 2. This zone is also characterised by the occurrence of a number of sparse macrofossils, which occur primarily at the bottom of Zone 2 in Core 1 between 53cm and 60cm depth and at the top of Zone 2 in Core 2 between 23cm and 27cm depth.

Finally, Zone 3 is dominated by *Bithynia leachii*; Foraminifera; Insecta; *Pisidium nitidum*; Ostracoda; and *Phragmites australis*; as well as the unidentified shell fragments. The narrowest zone in both cores, Zone 3 extends to the base of both cores from 60cm depth in Core 1 and from 41cm depth in Core 2. It contains the lowest total macrofossil abundance and richness of all the zones in Core 1. In Core 2, total macrofossil abundance is high due to the presence of unidentified shell fragments in this section of the core. The most common macrofossil in Core 1 is Insecta, occurring throughout the zone except for 62cm to 63cm depth and in Core 2 is Foraminifera, occurring between 42cm and 43cm depth.



5.3.2.2 Diatoms

Diatom abundance data were calculated from counts made from the lake core samples. In the 76 samples comprising Core 1, 94 diatom species were identified and in the 58 samples comprising Core 2, 59 diatom species were identified. In total, 97 diatom species were identified (Figure 5.13) which are differentiated between freshwater and freshwater-brackish, and whether their primary habitat niche association is pelagic, periphytic, epilithic or benthic (Kelly et al, 2005; Bennion et al, 2011; Algaebase, 2016). Core 1 is characterised by 35 (37%) benthic species (15 of which are shared with Core 2), 25 (27%) epilithic species (17 of which are shared with Core 2), 21 (22%) periphytic species (16 of which are shared with Core 2) and 13 (14%) pelagic species (8 of which are shared with Core 2). Core 2 is characterised by 18 (31%) epilithic species, 16 (27%) periphytic species, 15 (25%) benthic species and 10 (17%) pelagic species, and only contains 3 exclusive species: the freshwater pelagic species *Aulacoseira lacustris* and *Stephanodiscus parvus*; and the freshwater to brackish epilithic species *Diploneis interrupta*.

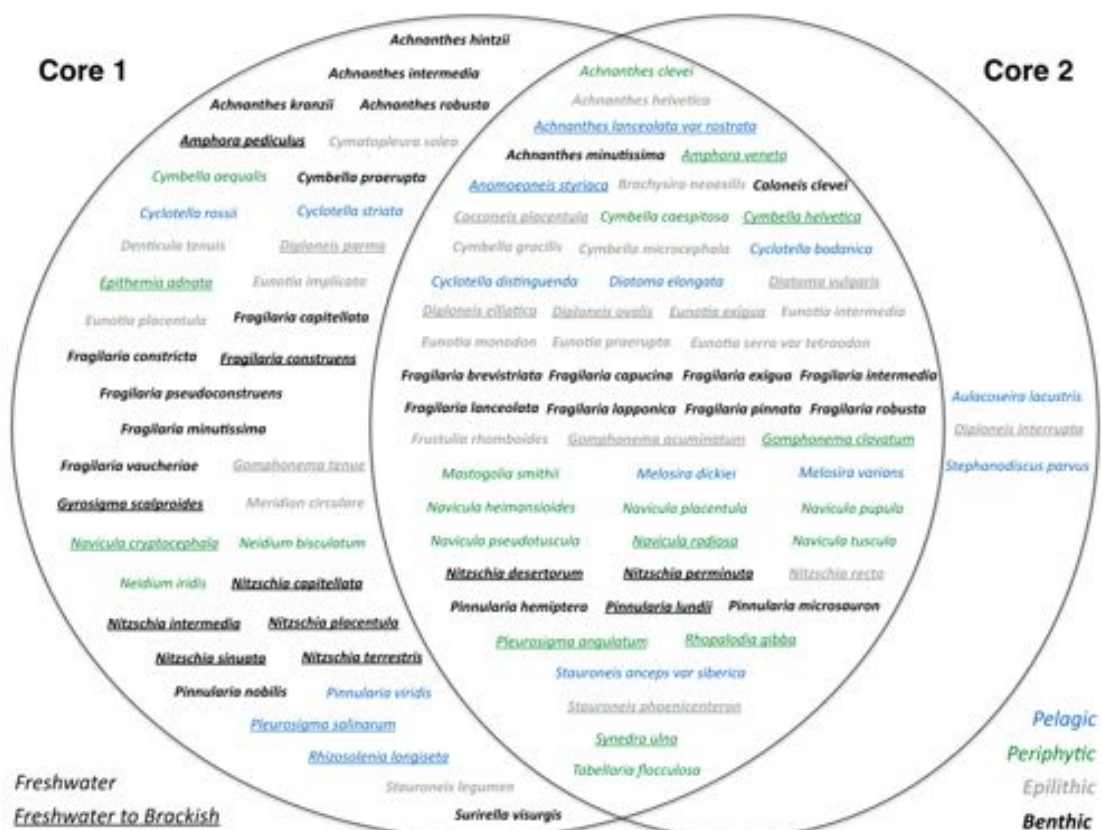


Figure 5.13 Venn diagram showing the 97 freshwater and freshwater-brackish lake core diatom species identified: only in Core 1 (38); only in Core 2 (3); and in both Core 1 and Core 2 (56). The diagram also indicates whether the primary habitat niche association is pelagic, periphytic, epilithic or benthic.

In both cores, the least abundant diatom species type are pelagic habitat-associated species; however Core 1 is dominated by benthic habitat-associated species, where as Core 2 is dominated by epilithic habitat-associated species. Both cores contain freshwater to brackish species (36% in both cores). In Core 1 and Core 2, the most abundant diatom is the benthic freshwater species, *Fragilaria exigua* (Figure 5.14). Similarly, the next most abundant diatoms in both cores are the periphytic freshwater species, *Tabellaria flocculosa* (17% in Core 1, 16% in Core 2), and the freshwater benthic species, *Achnanthes minutissima* (16% in Core 1, 11% in Core 2), along with the pelagic freshwater to brackish species, *Achnanthes lanceolata* var *rostrata*, which is also frequent in both cores (9% in Core 1, 6% in Core 2). Other frequent species include the freshwater benthic species, *Fragilaria intermedia* (5% in Core 1 and Core 2) and *F. brevistriata* (4% in Core 1, 3% in Core 2). Species that are frequent in Core 1 but not in Core 2 include the epilithic freshwater to benthic species, *Cocconeis placentula* (6%) and *Diatoma vulgare* (3%), where as the freshwater epilithic species *Cymbella microcephala* (5%) and *Achnanthes helvetica* (4%) are frequent in Core 2.

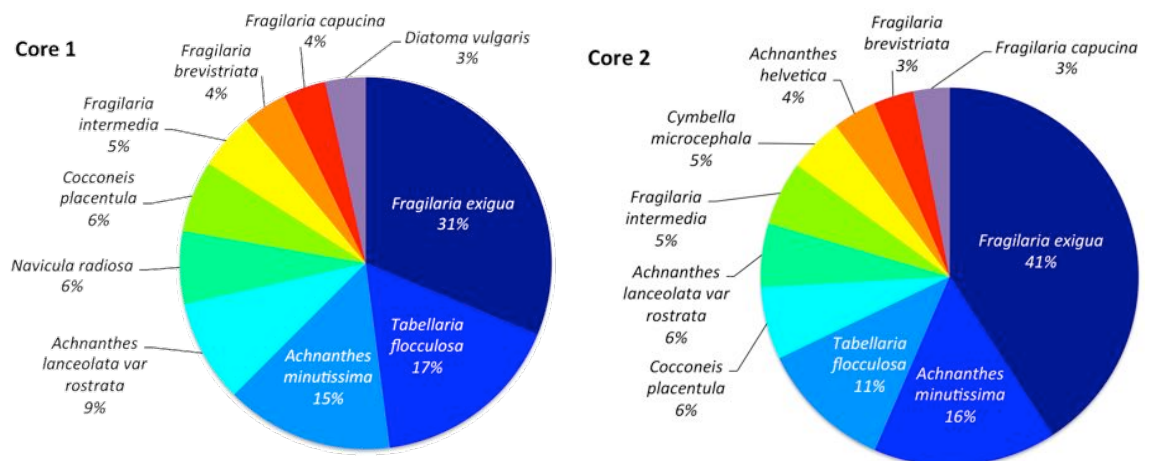


Figure 5.14 Pie charts showing the proportions of the 10 most abundant diatom species identified in the two lake cores extracted in June 2013.

TWINSpan cluster analysis was performed on the diatom data combined from both cores to identify communities and further assess the similarities and differences in these within and between cores. The TWINSpan communities are marked on the stratigraphic diagrams in Figure 5.15, which display the relative downcore diatom species abundance for both lake cores. Core 1 is characterised by all four communities where as Community 4 dominates the whole of Core 2.

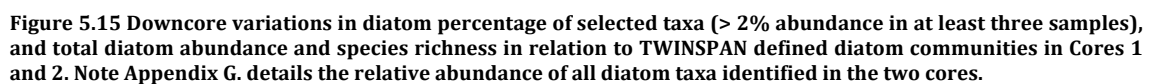
Community 3, present in Core 1 only, is dominated by the benthic freshwater diatom species *Achnanthes minutissima* and *Fragilaria exigua*, the latter of which increases in abundance with depth. Another dominant diatom is the periphytic freshwater species *Tabellaria flocculosa*, which decreases in abundance with depth. Other notable species include: the epilithic freshwater to brackish species, *Cocconeis placentula*, which increases in abundance with depth; the epilithic freshwater species, *Cymbella microcephala*, which peaks at the centre of this community section around 10 cm depth; and the benthic

freshwater species *Fragilaria intermedia*, which declines in the centre of this section. Less common species in Community 3 within Core 1 include: the pelagic freshwater to brackish species, *Anomoeoneis styriaca*; the epilithic freshwater species, *Cymbella gracilis*; the benthic freshwater species, *Fragilaria lanceolata*; the periphytic freshwater to brackish species, *Gonphonema clavatum*; and the pelagic freshwater species, *Melosira varians*. Community 3 extends from the surface to 18cm depth in Core 1 and is characterised by high total diatom abundance at the core surface to low total diatom abundance and high to moderate species richness at 18cm depth.

Community 4 on the other hand is found in both cores and is dominated by *Achnanthes minutissima*, *Fragilaria exigua* and *Tabellaria flocculosa*. Less common species include the benthic freshwater species *Fragilaria lapponica*, *F. minutissima*, *F. pinnata*, the epilithic freshwater species *Frustula rhomboides*, the epilithic freshwater to brackish species, *Gonphonema tenue*, the periphytic freshwater species, *Navicula placentula* and *N. pupula*, and the benthic freshwater to brackish species, *Nitzschia perminuta*. Community 4 represents the 19cm to 55cm depth section in Core 1 and the whole of Core 2 except for the barren samples near the core base. It is characterised by moderate to high total diatom abundance, high species richness. In Core 1, *Achnanthes minutissima* dominates this community, with *Fragilaria exigua* and *Tabellaria flocculosa* the next most abundant. In contrast, *Tabellaria flocculosa* dominates Core 2, although *Achnanthes minutissima* and *Fragilaria exigua* are also important. In Core 1 *Tabellaria flocculosa* is gradually exceeded by *Achnanthes lanceolata*, which increases downwards through the section. The epilithic freshwater species, *Achnanthes helvetica*, which is largely absent from Community 3, increases in the top half of this section in both cores.

The third defined diatom assemblage, Community 2, contains comparatively lower diatom species diversity. *Diatoma vulgare*, along with *Tabellaria flocculosa*, is the second most abundant species in this community, which is dominated by *Fragilaria exigua*. Other frequent species include *Achnanthes minutissima*, *Eunotia intermedia*, and *Navicula radiosa*. Rare species include *Achnanthes helvetica*, *A. lanceolata*, *Cymbella caespitosa*, *C. gracilis*, *Diatoma elongata*, *Eunotia exigua*, *E. intermedia*, *E. monodon*, *E. praerupta*, *Fragilaria brevistriata*, *F. capucina*, *F. intermedia*, *F. lanceolata*, *F. pinnata*, *F. robusta*, *Gonphonema clavatum*, *Navicula radiosa*, *Nitzschia desertorum* and *Nitzschia perminuta*. Community 2 spans the 55cm to 58cm depth section in Core 1 and is absent from Core 2. It is characterised by moderate total diatom abundance and species richness.

Similarly to Community 2, Community 1 contains high and moderate diatom species richness and abundance. This community, however, is dominated primarily by *Fragilaria exigua*, with *Tabellaria flocculosa* the second most abundant species. *Diatoma vulgare*, *Eunotia intermedia* and *Navicula radiosa* are also common, with *Achnanthes lanceolata*, *A. minutissima*, *Eunotia praerupta* and *Fragilaria brevistriata* present but rare. Unlike Core 1, which contains diatoms in all of the samples, Core 2 contains no preserved diatoms from 32cm depth to the bottom of the core at 46.5cm depth.



5.3.3 Stratigraphic evaluation

Visual assessment of the two lake sediment cores has revealed clear distinctions between the light and dark layering present within the two cores caused by alternations between light-coloured, minerogenic sand layers and darker more organic-rich peat layers. Also clearly identifiable within the two cores are the rapid transitions from sand-dominated sediments to peat-dominated sediments deposited in an environment separated from the influence of tidal sediment processes. Further stratigraphic analyses carried out on the cores reveal significant downcore variations in the sedimentology and palaeoecology, with distinct differences between the two cores.

Within Core 1, three sediment compositional stages are apparent (Figure 5.16). Firstly, the increase in organic matter and carbonate content with depth from the surface in the top two thirds of the core is reflected in a shift in mean grain size towards finer peaty sediments. Secondly, organic matter and carbonate content both peak at around 45cm depth, decreasing to very low levels between 55cm and 64cm depth, a pattern which is mirrored by the mean grain size which has the coarsest sand grain size between 45cm and 50cm depth. Thirdly, the organic matter and carbonate content at the surface is greater than the sediment in the top 20cm of the core, however the mean silt grain size at the surface is relatively fine. In contrast to Core 1, Core 2 displays a similar, but albeit less pronounced, pattern of three main downcore compositional changes. Firstly, the organic matter and carbonate content in Core 2 are both consistently low in the top 25cm of the core. This is reflected in the mean grain size, which is moderate through this section, except at around 17cm depth where fine sediments dominate. Secondly, there is a peak in organic matter in the peat layer between 25cm and 30cm depth; however this is not strongly reflected in the sediment grain size, which remains fine sand-sized. Thirdly, the bottom section of the core is characterised by low to near absent organic matter, which is paired with a surge in carbonate content and grain size that shifts towards fine silt-sized material.

The diatom communities identified in the two cores reveal a lake sequence that is dominated by freshwater-specific species, most notably benthic taxa *Achnanthes minutissima* and *Fragilaria exigua* (the most abundant species in both cores), and the periphytic species, *Tabellaria flocculosa*. In both cores, a downwards decrease in *Achnanthes minutissima* from the core surface is mirrored by an increase in *Fragilaria exigua* which dominates the sandy section near base of Core 1 and the bottom of the diatom section in Core 2 where diatoms are at their least abundant and species richness is low. Both cores contain a similar proportion of freshwater and freshwater to benthic diatom species; however Core 1 is dominated by benthic taxa whereas Core 2 is dominated by epilithic taxa. Diatom taxa primarily associated with pelagic habitats are by far the least abundant species type in both cores. The downcore distribution of diatom communities, however, differs between cores. Community 4, which contains the most even diatom diversity as *Fragilaria exigua* abundance is suppressed in this section, dominates the central portion of Core 1, and is associated with the whole of Core 2 except for the

barren samples from 32cm depth downwards.

By overlaying the macrofossil zones onto the stratigraphical summaries in Figure 5.16, it is clear that there is no direct alignment between the macrofossil groupings and the diatom communities within or between the two cores. The macrofossils identified from the lake sediments do, however, reveal three clear distinctions present in both cores. Firstly, the top sand-dominated section of both cores (0cm to 28cm depth in Core 1, and 0cm to 24cm in Core 2) is characterised by low diatom abundance but high species richness. Freshwater submerged charophytes, most notably *Chara spp.* and *Nitella spp.*, dominate this section along with the upland moss *Scorpidium scorpioides* and the freshwater cyanobacteria, *Gloeotrichia*. It should also be noted in this section of Core 1, the rare submerged macrophyte, *Najas flexilis*, is present in very low numbers from 2-12cm depth. Secondly, the middle peat section of both cores is dominated by *Phragmites australis*, from 28cm to 59cm in Core 1 and from 24cm to 41cm in Core 2, which mirrors the increase in organic matter. Thirdly, below the zone of *Phragmites australis*, in Core 1 there are no selected macrofossils present, except for the *Chara spp.* oospores identified at the base of Core 1 at 65cm depth. This zone in Core 2 is characterised by no diatoms present from 33cm depth downwards and a large rise in carbonate content to the base of the core from 40cm depth, reflecting the abundance of shell fragments in this section. In Core 1, however, carbonate levels largely decline and there is a significant increase in the abundance and dominance of the freshwater benthic diatom, *Fragilaria exigua*, and also the freshwater periphytic diatom, *Tabellaria flocculosa*.



5.3.4 Core chronology

Radiocarbon dating was performed by the Natural Environment Research Council (NERC), Radiocarbon Facility (NCRF) at East Kilbride in Scotland, initially on two sub-samples taken from the Core 1 samples A17 and A67, at depths of 12-13cm and 59-60cm, respectively. These samples represented the top and base of the peat-dominated layers within the core, and were deemed sufficiently organic for dating whilst achieving a good coverage in the stratigraphy. Dating revealed two estimated ages for the sediment in Core 1 (Table 5.1). The deeper sample, A67 at 59-60cm depth, yielded a date 353 ± 37 BP (SUERC-57619) (calAD 1472-1630), while the shallow sample, A17 at 12-13cm depth, produced the date of $3,398 \pm 39$ BP (SUERC-57710) (calBC 1643-1742). These ^{14}C dates were not as expected in terms of both age and stratigraphic order, especially given there were no problems during sample preparation and quality control data were within acceptable limits (see Appendix H. for the full NRCF dating report). Microscopic analysis of samples by NRCF revealed small dark fragments in both samples that were possibly coal fragments, and that these were significantly greater in frequency in the near-surface (A17) sample. Coal is resistant to the acid-alkali-acid pre-treatment, and its presence, even in small quantities, forces a shift to older ^{14}C dates due to the significant geological age of carbon contained in coal. It is possible that coal fragments entered the lake via aeolian deposition, having been carried by wind from the chimneys of neighbouring villages where solid fuel is commonly burnt.

Table 5.1 Radiocarbon (^{14}C) dating results for Core 1 [calibration performed by CalPal¹ and Calib 7.1²].

Sample	Elevation (mOD)	Depth (m)	1.0.1.1 Publication code	Age [conventional] BP* $\pm 1\sigma$	Age [calibrated ¹] date AD/BC	Age [calibrated ²] date AD/BC**
A17	2.49-2.50	0.12-0.13	SUERC-57710	$3,398 \pm 39$	1696 ± 45 calBC	calBC 1643 (1695) 1742
A25	2.44-2.45	0.17-0.18	SUERC-61822	632 ± 35	1338 ± 39 calAD	calAD 1294 (1349) 1390
A63	2.05-2.06	0.55-0.56	SUERC-61823	307 ± 35	1569 ± 54 calAD	calAD 1520 (1563) 1644
A67	2.01-2.02	0.59-60.0	SUERC-57619	353 ± 37	1543 ± 63 calAD	calAD 1472 (1549) 1630

* BP (before present) refers to conventional radiocarbon years before 1950 calAD

** median probability dates within one sigma (1σ) age ranges: calibration conducted using IntCal13

Radiocarbon analyses were therefore undertaken on two further sub-samples from Core 1 at 17-18cm (A25) and 55-56cm (A63) depths. These samples retained a good depth range within the core, but extended further into the peat-dominated section. Analysis of these samples yielded a date 307 ± 35 BP (SUERC-61823) (calAD 1520-1644) for the deeper sample A63 (55-56cm depth) and 632 ± 35 BP (SUERC-61822) (calAD 1294-1390) for the shallow sample A25 (17-18cm depth). Again, these dates present a range of issues and concerns when establishing a chronology for the sedimentary sequences examined. Shaw & Carter (1994) presented a date of 1040 ± 70 BP (calAD 894-1118) (Beta-22238) for the base of a sedimentary core from the margins of Sheskinmore (close to the Duvoge valley). Stratigraphic analysis showed evidence of a shift from estuarine to freshwater conditions

at that time, at an elevation of c. 1.4mOD. This has been interpreted as an approximate date for the initiation of lacustrine conditions at Sheskinmore. Given that other backbarrier coastal lakes in west Donegal have formed within a similar time frame (Shaw & Carter, 1994), this imparts some degree of control on the additional dates acquired here. Within this context, the mid-16th century dates obtained near the base of Core 1 (at 2-2.1mOD) work within the constraints of the pre-existing chronology, but the oldest date obtained from near-surface is substantially older than the system itself. The second near-surface date, despite existing within the time frame of the lake, is still problematic due to its inverse chronology and might be explained by contamination from the neighbouring layers.

Accepting that the near-surface samples were contaminated, and that there is no reason to doubt the chronology provided by the bottom samples, the sediment and ecological down-core analyses are presented within this dated stratigraphical context (Figure 5.17). Samples at A63 and A67 mark a distinct shift that is characterised by a fining upward sequence into increasingly organic, macrofossil-rich sediment and a change in diatom community. The paleoecology at the base of the core (c. early 1500s) is characterised by an assemblage (diatom Community 1) dominated by the benthic freshwater species, *Fragilaria exigua*, with the periphytic freshwater species, *Tabellaria flocculosa*, as the second most abundant species. The epilithic freshwater to brackish species *Diatoma vulgaris* is also common at this depth, as is the epilithic freshwater species *Eunotia intermedia*. The sample associated with the deeper radiocarbon date obtained (calAD 1472-1630) marks a shift in diatoms to an assemblage where the benthic freshwater species *Achnanthes minutissima* rises in prominence and *Fragilaria exigua*, though still present, decreases in abundance (Community 2). The epilithic freshwater to brackish species, *Diatoma vulgaris* is also abundant at this depth; however the only other important diatom at this depth is the periphytic freshwater species *Navicula radiosa*.

In comparison to diatoms, macrofossils at depth are sparse, but up-core from 60cm depth (calAD 1472-1630) *Phragmites australis* and other types including *Gloeotrichia* spp. and *Scorpidium scorpioides* become increasingly abundant. There are local peaks in freshwater cyanobacteria *Gloeotrichia* spp., the common reed *Phragmites australis*, and the upland moss *Scorpidium scorpioides* a couple of decades later at 55cm depth (calAD 1520-1644) where total macrofossil abundance reaches a core maxima. Up-core to the surface sees a period of dominance by *Phragmites australis* coupled with the diatom species, *Fragilaria exigua*, and a complete decline in *Eunotia intermedia* and loss of *Diatoma vulgaris*. Throughout this period (30cm) of *Phragmites* dominance, which likely reflects the mid-1500s to the late 1700s, there is also a notable increase in *Scorpidium scorpioides*, along with two significant phases of expansion and decline in *Gloeotrichia* spp., *Nitella* spp., *Chara* spp. and *Daphnia ephippia* which each peak around 55cm and 40cm depth. At 30cm depth, *Phragmites australis* completely disappears from the sediment record and following this, there is an increase in the diatom species *Achnanthes minutissima* and *Tabellaria flocculosa*, which overtake the declining *Fragilaria exigua* to become the dominant taxa.

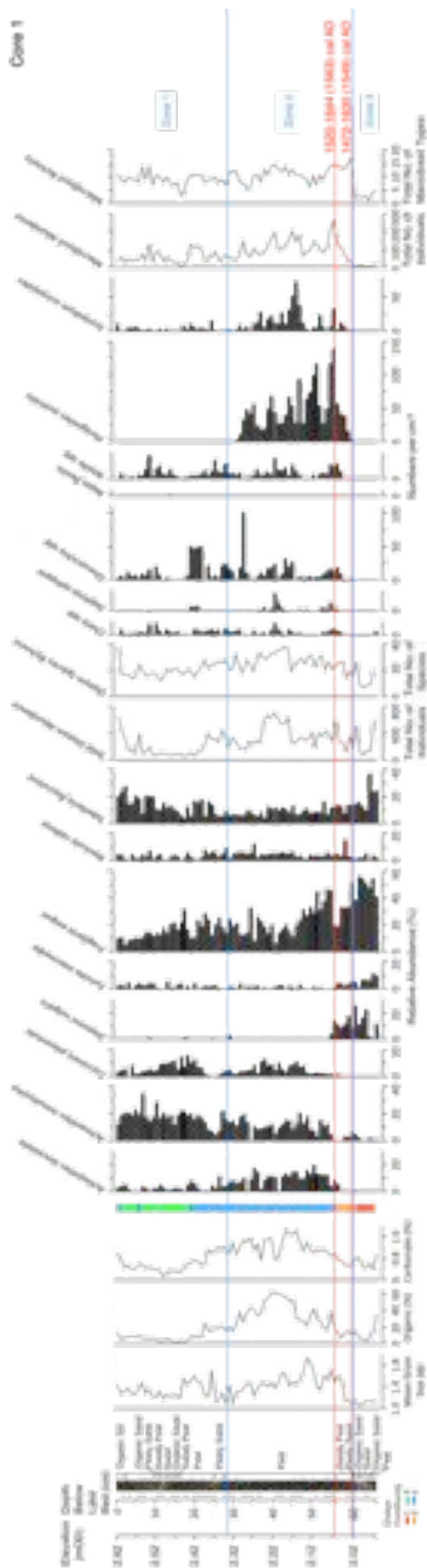


Figure 5.17 Summary of downcore profiles in Core 1 with CalPal calibrated ¹⁴C radiocarbon dates overlain.

5.3.5 Historical context

To place the stratigraphic data and chronology into a spatial history context, historical maps, aerial photographs and satellite images were compiled and examined. The earliest mapping evidence of the Sheskinmore Lough area is from the 1700s (Figure 5.18). The key observation from both maps (1775 Murdoch MacKenzie and 1777 George Taylor / Andrew Skinner) is that they do not show Sheskinmore Lough. Importantly, Lough Fad is marked on the 1777 map; this lake currently covers an area of 0.32km² in comparison to an area of 0.37km² for Sheskinmore Lough. In the place of Sheskinmore Lough is a river connecting Lough Fad to the Loughros More estuary via the site where Sheskinmore Lough is now present. The absence of the lake from the 1775 map is less surprising, however, given its cartographical simplicity and the absence of lake features across the wider area. In addition, the James McParlan Statistical Survey of The County Of Donegal 1801 map also does not show Sheskinmore Lough (Figure 5.19).

Sheskinmore Lough is clearly defined as a lake on maps and charts from the early 19th century (Figure 5.20). In 1835 it is depicted extending much further up the Duvoge river valley northeast towards Lough Fad than the current lake. Although the river from Lough Fad is not shown connected to Sheskinmore Lough in this map, the depicted river sections do closely follow the present-day Duvoge river that connects the two lakes. Interestingly, an additional inflow from Summy Lough is depicted flowing into this northeast arm of the lake. Presently, the course of this stream lies further northeast joining the Duvoge river at a confluence further upstream towards Lough Fad. The other feature to note from the 1835 map is that the surface area of the lake is more significant that at present, in both a northwesterly and a southerly direction. This is also reflected on the 1860 map.

In contrast to the 1800s, maps from the early 1900s show a smaller lake area bordered from the west to the northeast by wetland, rather than open water (Figure 5.21). The 1901 map depicts a smaller open water area in comparison to the wetland area, which is concentrated towards the northern side of the lake. Similarly, the 1905 map shows a much reduced lake area, however, the open water is distributed primarily to the western and southern side of the lake area outline. These wetland areas are all annotated with 'Liable to Floods' suggesting the lake area intermittently increased in size during wetter conditions. In addition, a date and elevation is marked on the lake - 'Surface of Water 19.4 12th December 1903', which equates to 3.21 mOD (the lake surface at 19.4 ft is referenced to Poolbeg Datum, which is 2.7 m above the Malin Head Datum or sea level).

Aerial photographs from the mid to late 1900s reveal more detail about Sheskinmore Lough and the surrounding area that were not necessarily captured in the earlier cartographic maps (Figure 5.22). In these three time frames (1951, 1977 and 1995), the channel feature can clearly be seen extending across the lakebed from the northeast corner (the Duvoge River inflow) to the southern side of the lake. There is no suggestion through these images that the channel has changed in planform or morphology, indicating that it is a relic feature that has not been dynamically forced or modified in recent decades. Across the wider wetland, boundaries enclosing strips of land running north-south had

been introduced by the 1950s, and through to the 1990s there are some differences in the vegetation signature here, perhaps expressing changes in grazing practices.

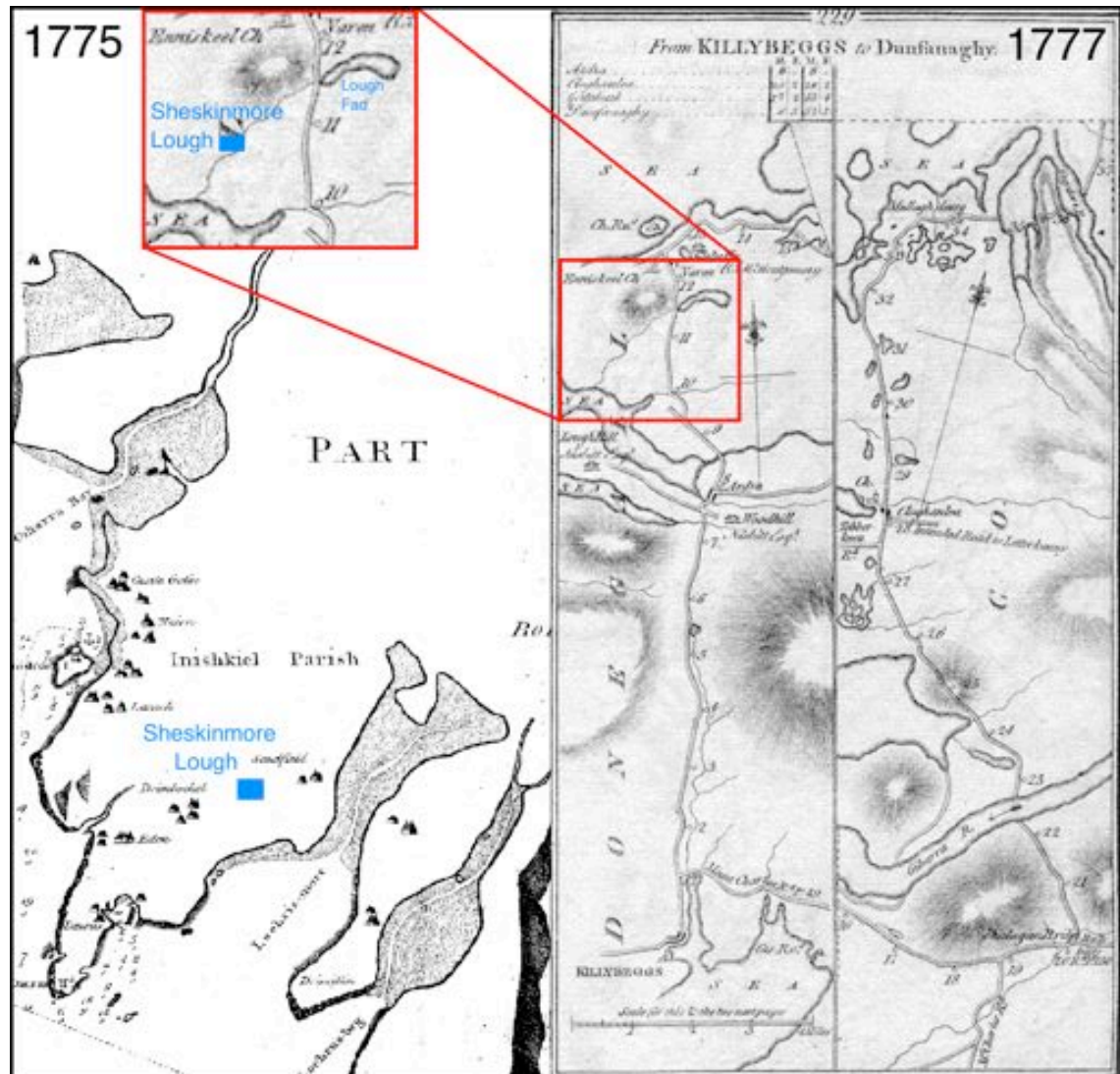


Figure 5.18 The 1775 Murdoch Mackenzie map of the Inishkeel Parish and the 1777 George Taylor & Andrew Skinner map, the left panel of which marks the main road north from Killybegs to Inishkeel via Ardara (MacKenzie, 1775; Taylor & Skinner, 1778). Note that no lake is recorded on either map in the vicinity of Sheskinmore Lough, so the approximate position has been marked in blue.

Significant changes in the sand dune system are also evident over this timescale. The brighter signal in the 1951 image implies lack of vegetation and aeolian activity across the dunes, and there is some evidence to suggest that the early 1900s was characterized by an increased storminess, which led to destabilization in the dune system here and elsewhere in west Donegal (Burningham, 2008; Bolles, 2012). The gradual darkening of the aerial imagery through 1977 to 1995 represents a reduction in aeolian activity and increase in vegetation cover, where erosion and sediment dynamics become increasingly confined to blowout areas. Throughout this period, the southern margin of the dunes has eroded significantly, driven by northward migration of the low tide channel (Burningham, 2008). Closer to the lake, there appear to be patches of wind blown sand deposited on the

western and southern margins in all the images. In addition, the 1977 image shows similar deposits around the mouth of the Duvoge river to the northeast of the lake and on the raised area of peatland that separates the Abberachrin and Duvoge rivers to the east of the lake.

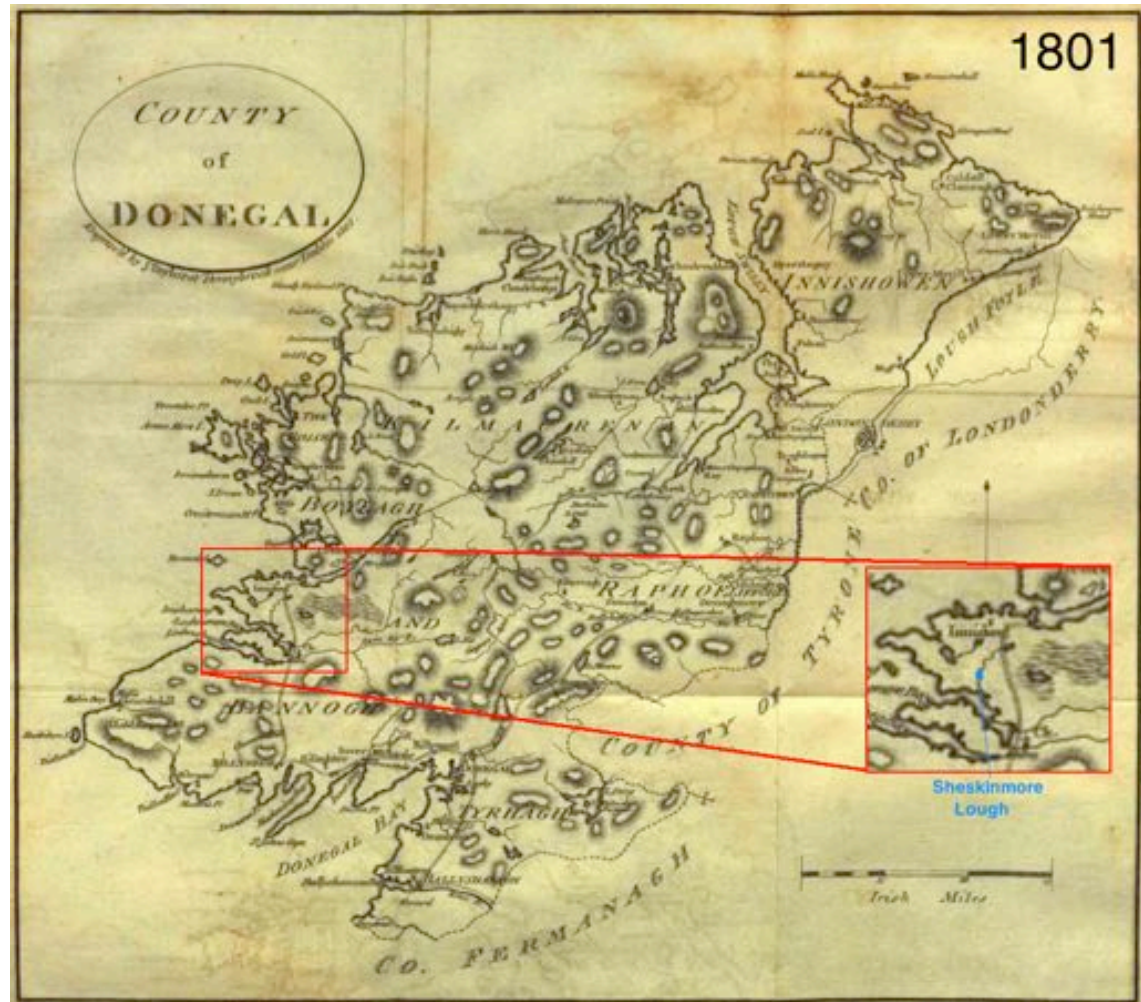


Figure 5.19 The 1801 James McParlan Statistical Survey of The County Of Donegal map (McParlan, 1802). Note that no lake is recorded on the map in the vicinity of Sheskinmore Lough, so the approximate position has been marked in blue.

Colour imagery from 2000, 2005 and 2012, reveals how the estuary meander has continued to erode this section of the dunes, albeit at a slower rate (Figure 5.23). When comparing the 2000 and 2005 images, there is clear development of vegetation in the present-day reedbed area to the northern and western sides of the lake. In the 2005 image the areas of vegetation, most likely dominated by *Phragmites australis*, appear to be more continuous and the areas of open water in this vegetation-dominated zone are more reduced in size. Caution should be applied, however, when making this observation as the lack of ponding in the dune blowouts and the smaller surface area of the surrounding lakes in comparison to the 2000 and 2012 images suggests the 2005 image was captured during a dry period when surface water levels were lower than average. But features such as the within-lake channel system show no evidence of change over this time frame.

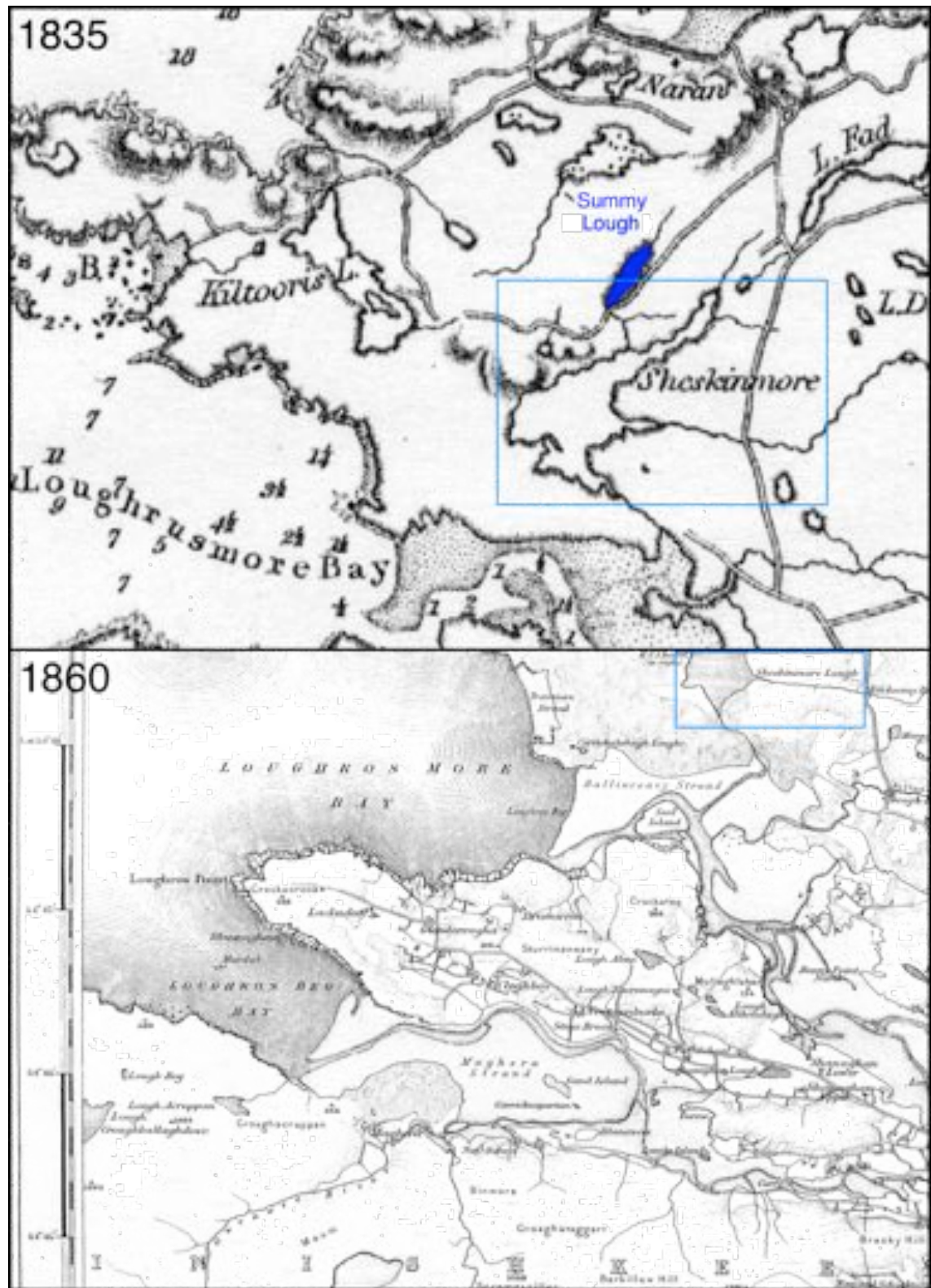


Figure 5.20 Section of sheet V, Farland Point to Donegal Bay, from the 1835 UKHO admiralty chart of Ireland surveyed by Captain Mudge, and a section of the 1:25,000 first edition of the '1 inch to 1 mile' OS map from 1860, both held in the British Library (Burningham, 1999). Sheskinmore Lough is outlined in blue.

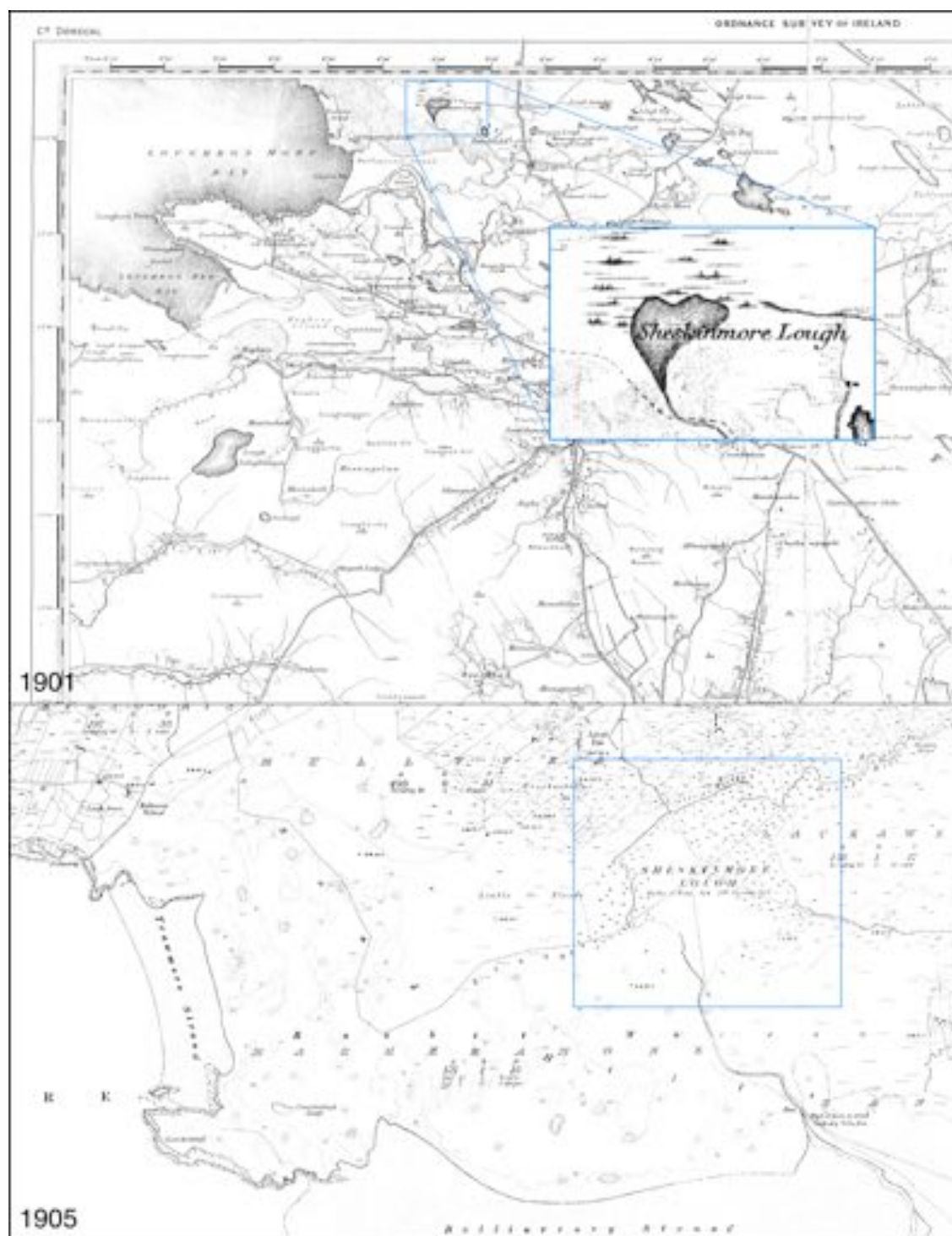


Figure 5.21 Section of the second edition of the '1 inch to 1 mile' OS map at 1:25,000 scale from 1901, and a section of the second edition OS 1:10,560 map published in 1907 but based on revisions conducted in 1905 (Burningham, 1999). Sheskinmore Lough is outlined in blue.

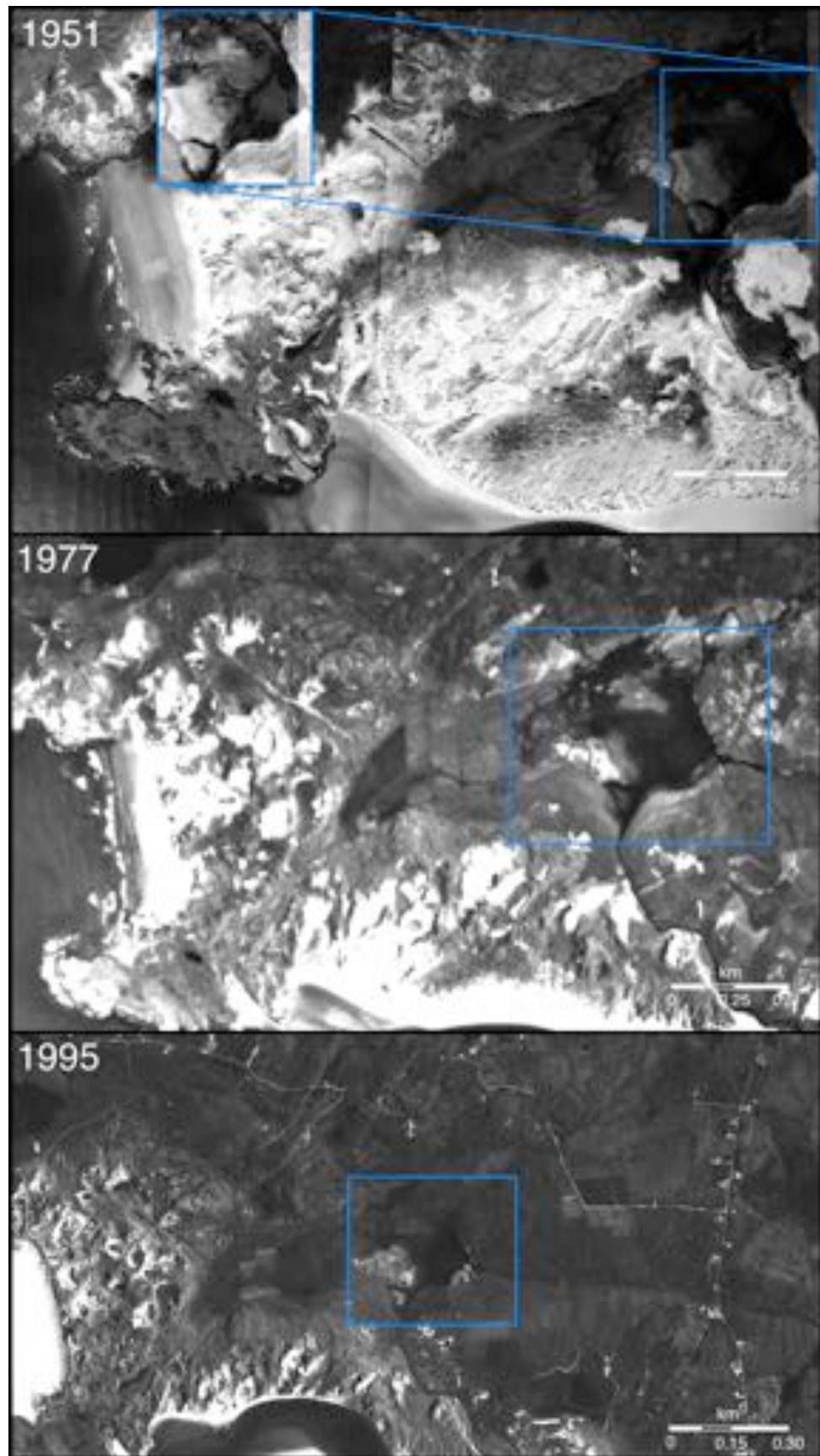


Figure 5.22 Sections of aerial photographs from 1951 (Irish Air Corps), 1977 (Ordnance Survey of Ireland) and 1995 (Ordnance Survey of Ireland) (Burningham, 1999; Ordnance Survey Ireland, 2016). Sheskinmore Lough is outlined in blue. Note the exposure and contrast of the Sheskinmore Lough in the 1951 photograph has been edited to highlight the lake bed, shown in the inset.



Figure 5.23 Satellite images 1:10,000 from 2000, 2005 (both from Ordnance Survey Ireland) and 2012 (from Bing) showing Sheskinmore Lough at the centre of each image (Bing, 2016; Ordnance Survey Ireland, 2016).

5.4 Discussion

Sheskinmore Lough has undergone significant spatial and temporal ecohydrological and environmental changes over time. The lithostratigraphic and paleoecological results point to three key phases in the evolution of the ecohydrological and environmental conditions of the lake and wetland system. Before examining those time periods in more detail, it should be noted that the core sediments do not capture the shift from estuarine (when seawater had access to the site) to freshwater conditions that were identified in Shaw & Carter (1994). This is due to the focus on shorter lake cores in the present study. Shaw & Carter (1994) described estuarine sediments from a depth range of 182–200cm at c. 1.2–

1.4 OD (Table 5.2). Above this transition, they describe herbaceous peat, and interestingly no evidence of sand-dominated layers. This is likely due to the location of the Shaw & Carter (1994) core at the mouth of the Duvoge river floodplain, which is out of the range of aeolian-driven activity and perhaps too marginal to respond significantly to possible shifts in open water extent.

Table 5.2 Description of the lithostratigraphy and paleoecology in the Shaw & Carter (1994) core extracted from the wetland to the northeast of Sheskinmore Lough.

Depth Range (cm)	Lithostratigraphy	Paleoecology
0 - 182	Very dark brown poorly humified herbaceous peat.	<i>Phragmites</i> rhizomes and stems.
	Fine disseminated sand.	<i>Betula</i> fragments at 60cm and 150cm.
182 - 200	Very dark greyish brown fine sand.	Monocot roots.
	Disseminated mica.	Some organic material.

Although the marine to terrestrial transition was not captured in this study, the cores reveal important climatic and geomorphological shifts that define the three key phases in the recent environmental and ecohydrological history of the lake and wetland system. Specifically, this comprises a shift in the 1500s from a sandy environment to one dominated by peat, followed by an increase in flooding and evolution to a lacustrine environment. These phases are summarised in Figure 5.24, which depicts the main geomorphological and hydrological characteristics of the Sheskinmore Lough area.

Phase 1 (? to calAD 1472-1630): prior to the mid-1500s, the Sheskinmore Lough environment was characterised by an active sandy environment where vegetation cover was relatively limited, conditions were possibly dry and windy, driving aeolian sediment transport and deposition leading to hummocky terrain across the area landward of the dune system. The low diatom and macrofossil abundance and diversity, the presence of various shell fragments and the lack of very fine sediments indicate that the aquatic system was limited, sediment was mobile, vegetative productivity low and the environment was likely relatively dry beyond the Duvoge and Abberachrin rivers. It is unclear what path these rivers took at this time, though they were likely more dynamic and responded to shifts in morphology associated with aeolian processes. It is also unclear how the system evolved between the separation from the estuary around 1000 calAD (calAD 894-1118; Shaw & Carter, 1994) and the 1500s, and to what extent a lake occupied the site during this period.

Phase 2 (calAD 1472-1630 to c.1800 calAD): by the late 1500s, Sheskinmore Lough area entered a wetter period that is evidenced by decreased sand mobility, vegetation growth, peat accumulation and a substantial shift in aquatic flora. The ecohydrology of the system is characterised by an increase in diatom and macrofossil abundance and diversity, the latter of which is dominated by *Phragmites australis*. There is limited evidence of lacustrine features during this period, and it is more likely that Sheskinmore comprised an extensive wetland system where significant stands of *Phragmites australis* lined the Duvoge and Aberarachrin rivers as they crossed the lowland northeast of dune system. This phase persisted for up to

250 years, during which time there were shorter ecohydrological cycles that forced shifts in diatom and macrophyte communities, possibly reflecting changes in nutrients and trophic status.

Phase 3 (c.1800 calAD to present): into the early-1800s through to present, a lake system has occupied the Sheskinmore site. Historically, this is evidenced on maps, but also in the lake sediments that show a distinct shift from reedbed to an open water aquatic environment with an associated decline in *Phragmites australis* and reduction in organic deposition. This implies a further shift to a wetter climate, but might also express anthropogenic influences in the wider catchment. Stratigraphic evidence shows that the last 200 years have experienced some fluctuation in environmental conditions, most notably evidenced by the successive, but episodic layers of aeolian sand deposits within the accumulating peaty sediment. The ecohydrology of the system during this period is characterised by several small-scale shifts in diatom species dominance, with an overall decline in *Fragilaria exigua* and *Cocconeis placentula* and increase in planktonic *Tabellaria flocculosa*.

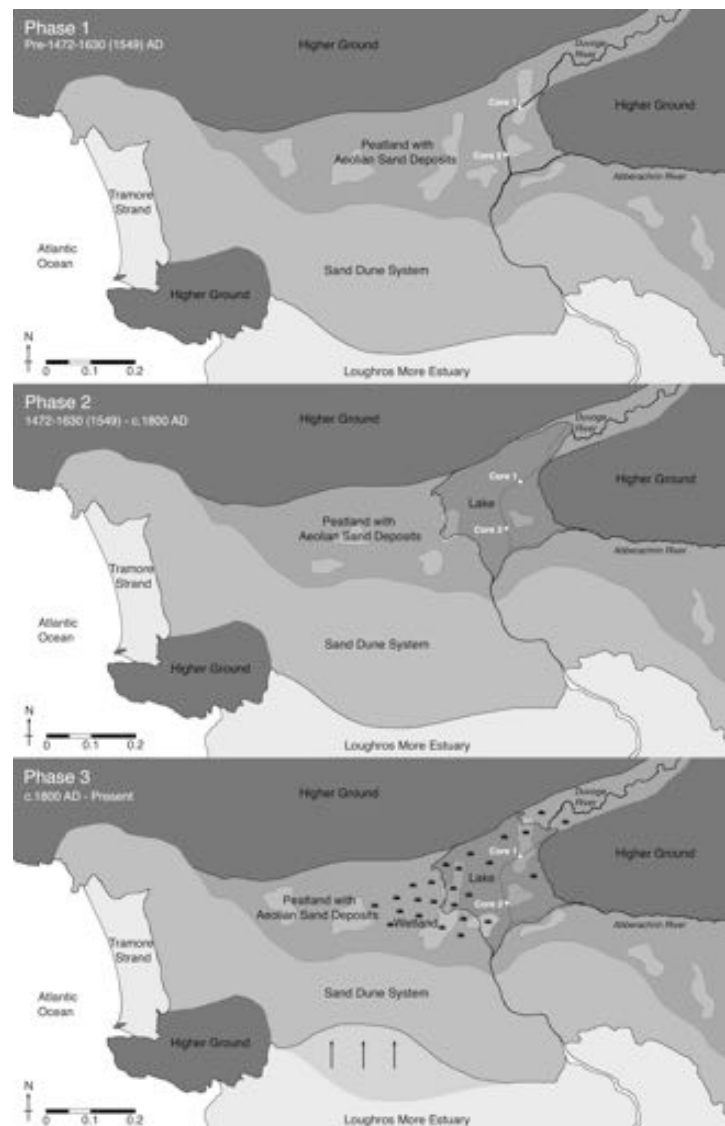


Figure 5.24 Stages in the development of Sheskinmore Lough from the 1500s to present.

Much of the history represented in the Sheskinmore Lough cores covers the period influenced by the Little Ice Age (LIA) (1570-1900 AD; Clarke & Rendell, 2009), when average temperatures across Europe were 2°C lower than they are today, and Ireland experienced a long period of wet conditions (Swindles et al, 2013). The LIA was also responsible for long cold winters in Ireland, with temperatures not reaching much above freezing at the end of the seventeenth century (Pauling et al, 2006; Kiely et al, 2007). Summer precipitation across Europe from 1500 AD to the mid 1600s is characterized by a gradual increase in rainfall throughout the period (Pauling & Paeth, 2006; Pauling & Paeth, 2007). The stratigraphic evidence from Sheskinmore captures the transitional decades prior to the onset of the LIA (Phase 1), the period of the LIA (Phase 2 and the start of Phase 3) and of course the century since its cessation (much of Phase 3), all of which is displayed in Figure 5.25.

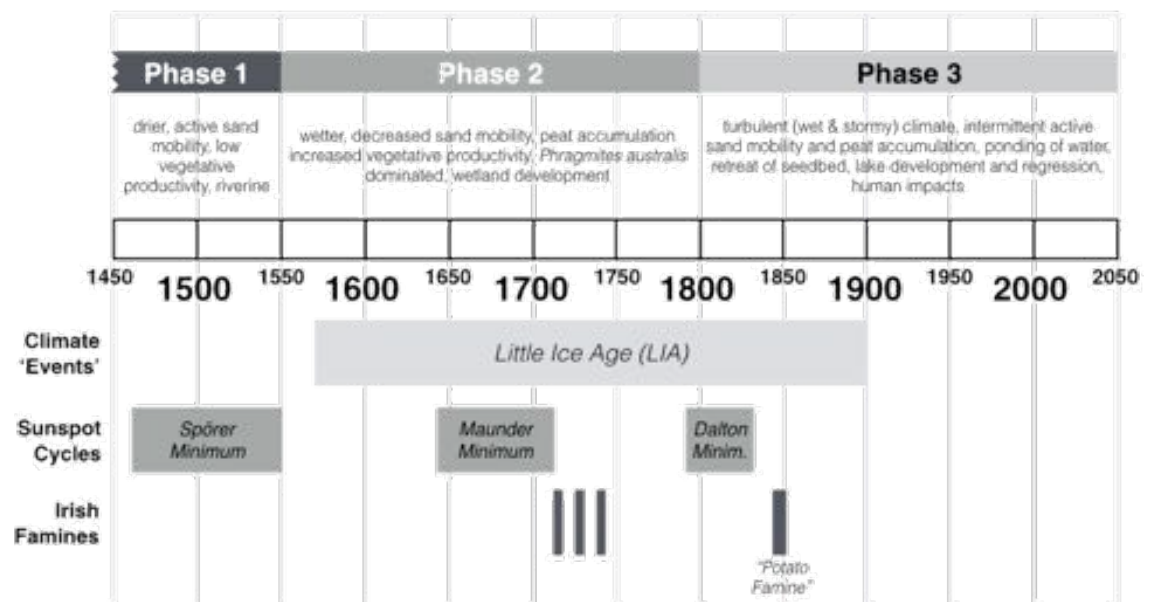


Figure 5.25 Timeline summarising the historical phases identified from the stratigraphic record in relation to key climate events, sunspot cycles and Irish famines.

The inference of aeolian processes evident in the lake core lithostratigraphy prior to 1472-1630 (1549) AD indicates a windy climate, and the lack of peat formation suggests a lower frequency of wet years on average in *Phase 1* when compared to *Phase 2*. Several studies investigating historical documentary evidence have shown that the early 1500s were characterised by stormy weather with an increased frequency of cyclonic storms in the north east Atlantic and several accounts of major storms occurring around British and Irish coasts (Marusek, 2010; Rowley, 2016).

Stratigraphic evidence shows that *Phase 1* can be characterised by freshwater clear oligotrophic conditions with a circumneutral to slightly alkaline pH. The benthic diatom species, *Fragilaria exigua* dominates. High relative abundances of *Fragilaria spp.* are typical of clear, alkaline, shallow freshwater bodies of intermediate nutrient status where a favourable light climate supports growth on the surface sediments or attached to plant surfaces (Bennion et al, 2011). *Fragilaria exigua* numbers are at their highest abundance within the sand layer at the base of Core 1, indicating water levels were low and carbonate

shell material, present within the wind blown sediment, had an impact on the hydrochemistry at the core site (Leira et al, 2007). Denys (1988) documented that *Fragilaria* taxa are typical of shallow, disturbed water environments, with Van de Vijver & Beyens (1999) observing that *Fragilaria* taxa generally dominate river, rather than lake, diatom flora. In addition, the benthic and epilithic species that characterise this phase suggest water velocity had an impact on the resulting diatom assemblage (Ni Chathain et al, 2004). Indeed, in northern ecoregions Potapova & Charles (2002) observed that epilithic species prefer fast-flowing cold waters indicative of riverine habitats and also argue that water velocity is an important factor determining the distribution of benthic species in aquatic systems.

The diatom assemblages of this phase support the idea that the river would have flowed through peatland, much like the modern floodplain of the Duvoge river, which would have been subjected to aeolian sand deposition (Ghosh & Gaur 1998; Antoniadou & Douglas, 2002; Massard & Geimer, 2008; Soininen & Eloranta, 2016). The exploratory wetland cores and those from the lake demonstrate the significant variation in the distribution and depth of sand layers across the site. Aeolian deposition would not have been uniform across the site, and a hummocky terrain more likely.

The pre-lake state is also supported by the low abundance and diversity of macrofossils in the lowest part of the core. Macrofossils are almost completely absent from *Phase 1* (i.e. in the decades before the LIA). There is evidence of the freshwater charophyte, *Chara aspera*, in the peaty base, but *Chara spp.* disappears in the short sequence above this, prior to the mid-1500s. Even Insecta, which are likely to be a mixture of terrestrial and aquatic species, are absent in the centre of this phase, suggesting widespread disturbance that comprised frequent wind-blown sand movement and reduction in vegetation growth. In addition, given only marginal wetland (e.g. *Equisetum fluviatile* and *Juncus bulbosus*) and terrestrial meadow macrofossils (e.g. *Eupatorium cannabinum* and *Plantago lanceolata*) are present, and in very low numbers, the site was more likely to be characterised by a river floodplain environment that may have flooded from time to time, during this pre-LIA period. Interestingly, the name Sheskinmore Lough (Loch an tSeascainn Mhóir in Irish) translates thus: *seisceann* (a marsh, a boggy place); *mór* (big); *An tSeascainn Mhór* (a large marshy area) (McGill, 1992). Although McGill (1992) does not state when Sheskinmore Lough would have likely been named, the author does mention that the adjacent sandy plain of Magheramore (which translates as a big flat area) is widely documented in the 1600s as the location for a great annual fair or *aonach*, and also a much earlier tribal assembly point dating back to early Celtic times. It is therefore highly likely that Sheskinmore was named during Phase 1, or earlier, when the site was dominated by a river floodplain, rather than a lacustrine, environment.

Phase 2 can be characterised by clear freshwater oligotrophic conditions with a circumneutral to slightly acidic pH. This lowering of the pH in comparison to Phase 1 is in large part due to the accumulation of peat within the substrate. The river/floodplain environment of Phase 1 would have existed, but the rapid (over a decadal time-scale) increase in *Phragmites australis* and peat formation implies a shift to a colder, wetter time

period. On the basis of cartographic evidence, the wetter riverine conditions at Sheskinmore (Phase 2) continued for around 250 years to c. 1800 AD. This period lies firmly within the climatology of the Little Ice Age, central to which was the Maunder Minimum decline in sun spot activity (1645-1715 AD) (Blackford & Chambers, 1995). According to Hickey (2011), this period was characterised by colder and wetter than average conditions in Ireland. Using Irish blanket peat, Blackford & Chambers (1995) show that colder and wetter periods recorded in the peat at the western oceanic fringes of Europe can be linked to inferred variations in solar output over the last millennium. In particular they suggest that this fringing area is more sensitive to climatic change. Although evidence of the impacts of this climate change in northwest Ireland is limited, analyses from across Ireland show flooding and increased precipitation were associated with the LIA (Blackford & Chambers, 1995; Swindles et al, 2013). Marusek (2010) documents that flooding occurred in Londonderry in 1680 AD, in Dublin in 1687 AD, along with excessive rains across Europe and Ireland in 1705, and across Ireland in 1707 AD.

Post 1720s, this wet trend continued, with great rains and floods recorded on multiple occasions across Ireland between 1729 and 1777 (Marusek, 2010). Following one of the longest and severest winters in the eighteenth century from 1783 AD to 1784 AD, the melting of snow resulted in great floods across Ireland and the rest of Europe, and in 1787 AD there were great floods in most of the principal rivers in Ireland (Marusek, 2010). This wet climatic environment would have favoured the development of peatland across the backbarrier / river floodplain environment (overlying the sand-dominated stratigraphy associated with *Phase 1*) and was also key to shaping the hydrological processes that led to the formation of the lake by the early 1800s, marking the transition to *Phase 3*.

It is possible that the progressive development of the *Phragmites* wetland during the first half of the LIA brought about a sufficient change in the environment and hydrology that was then capable of forcing further change to a lacustrine system. Drainage is increasingly impaired during peat formation and accumulation, which can lead to pooling of water and formation of ponds and lakes within the wetland environment (Comas et al, 2004), but climatic controls are known to exert a clear influence on the development of open water conditions driven by shifts from drier to wetter conditions (Barber, 1981; Boatman & Tomlinson, 1973; Glaser & Janssens, 1986). Given the wettening climate during the 1600-1800s, and the rapid increase in *Phragmites australis* in the sediment record, it would seem that the formation of Sheskinmore Lough is primarily due to a shift towards wetter climatic conditions over this period.

Discrete periods of flooding interspersed with periods when the water level was below the ground surface can be identified within the sediment record during *Phase 2* through the aligned, but negatively correlated cycles of growth and decline in *Phragmites australis* and charophytes. Periods of increasing abundance of *Phragmites* coincide with declines in the abundance of the charophytes *Chara spp.* and *Nitella spp.* and vice versa, the latter being commonly associated with shallow standing rather than running water systems. The intermittent declines in charophyte abundance are also mirrored by declines in other lake-dwelling macrofossils such as *Daphnia ephippia*, *Isoetes lacustris* and *Nymphaea alba*. The

majority of peaks in *Phragmites australis* abundance are accompanied by peaks in the abundance of the moss, *Scorpidium scorpioides*, most commonly associated with the mineral-rich flushes of upland bogs (Atherton et al, 2010). Amsberry et al (2000) observed that low oxygen availability, characteristic of waterlogged soils, limits *Phragmites* growth. Whyte et al (2008) also observed that invasion of *Phragmites australis* is facilitated by declines in water lake levels. After the initial invasion by *Phragmites australis* when the climate became wetter and wetland environments developed along the river margins, the subsequent short-lived declines in *Phragmites australis* abundance may represent excessive flooding and pooling of water, with subsequent expansion then signalling periods of time experiencing some reduction in water level. With progressive flooding and an increase in water levels, *Phragmites australis* was gradually phased out in the centre of the Sheskinmore site. But the presence of *Phragmites australis* north and west of the coring site in the modern lake shows that these changes do not capture the full spatial context. Presence of *Phragmites* at the coring site implies that the area was choked by reedbed, but declines in abundance in the sedimentary record likely reflect retreat of the reedbed to broader margins with expansion of open water.

It is hard to say whether the disappearance of *Phragmites australis* from the sediment record during this phase marked the exact time the lake was formed as peat formation continues to occur after the species has largely disappeared from the sediment record; however it is clear its decline was most likely due to rising water levels. Cartographic evidence in 1835, when Sheskinmore Lough first appears on a map, points to a lake system that was much more extensive than present. It is possible that features are exaggerated on early maps; however the flooded Duvoge river valley, and lack of reedbed in maps from the 1800s imply a greater extent of open water than present, and certainly far more than 18th century maps and stratigraphic evidence. As previously noted, there is a strong possibility that the continuing climatic driver of the LIA provided the necessary wet conditions that facilitated the shift from wetland to lake. But a further possible explanation for the development of open-water conditions is through damming of the Duvoge and Aberachrin rivers, particularly in a location south of their confluence. This would likely have been forced naturally, a consequence of dune dynamics blocking the entry of the rivers to the Loughros More estuary (the precedence for this is in part provided by earlier phases of late-Holocene dune dynamics (see Wilson, 1990; Carter & Wilson, 1993; Shaw & Carter, 1994)), allowing water levels to rise.

A feature that perhaps adds some weight to this argument is the distinct meandering channel that crosses the lake evident in the aerial photography. The path from the confluence of the Duvoge and Aberachrin (along the bedrock margin of the interfluvial) crosses the lake and seemingly enters the hinterland of the southern margin: the sandy infill between Sheskinmore and Sandfield Loughs. The present-day geomorphology implies a blockage to this channel occurred at some point in the past. Indeed, this blockage may well explain the incised nature of the outflow channel, which is likely to have been created artificially in order to reconnect the river to the estuary because of excessive flooding upstream within the Duvoge valley (Emer Magee pers comm. 23rd June 2012).

Although entirely speculative, it is important to recognise that dune dynamics have been responsible for the development of coastal lakes across the west coast of Ireland and beyond through the rapid and large-scale redistribution of sand during stormier climate episodes.

There is potential evidence of fluctuating lake levels during *Phase 3* once an open water system had developed. The vast majority of macrofossils recovered from the upper part of the core are characteristic of a shallow lake environment, with some also providing evidence of marginal flora. For example, at c. 22cm depth the lake level at the core site is unlikely to have exceeded 1m depth due to the presence of the various-leaved pondweed, *Potamogeton gramineus* (Preston, 1995). *Najas flexilis* is variably present in the top 15cm suggesting that conditions for its success were only achieved around 100 years ago, and that these fluctuated over that time frame. *N. flexilis* is well documented in water depths of 1-5m (Pearsall, 1917; Sheldon et al, 1977) but its sensitivity to the combination of water depth and exposure (Wingfield et al, 2004) implies that these factors were variable over a decadal time-scale during the last century. During periods of its absence, species such as *Menyanthes trifoliata* (Han & Kim, 2006) and *Chara aspera* (Van den Berg et al, 2001) that succeed in shallow (<0.5m) water appear providing further evidence that water levels have fluctuated.

Phase 3 would have initially been characterised by the same cold and wet conditions in Ireland that occurred throughout the LIA. Exceptional rain and flooding was recorded across Ireland in the early 1800s, with 1815 AD named 'the year with no summer', and huge flooding and snowfall events throughout the 1810s and 1820s (Jeffers, 2014). These climatic conditions are a well-documented basis for the Irish Famine between 1845 AD and 1847 AD, and also presented Ireland with its first documented significant storm in 1839 AD (Austin Bourke, 1965; Cole & Mitchell, 2003). Dubbed the 'Night of the Big Wind', this remains was one of largest recorded storms in Ireland's history (Shields & Fitzgerald, 1989). Although the effects of the Little Ice Age were coming to an end during the second half of the 19th century, the last cold snap occurred between 1878 AD and 1898 AD. The beginning of the 20th century, in comparison, was dominated by more unsettled climatic conditions across northwest Europe (Clarke & Rendell, 2009). This can be inferred from the sedimentary record where peat is interspersed by an increasing number of aeolian sand layers that are likely to have been deposited during stormy conditions. Significant storms are recorded in 1903, 1917 AD and 1927 (Lamb, 1991), and the North Atlantic Oscillation (NAO) has provided further corroboration of increased storminess during these early decades of the 20th century (Lozano et al, 2004). The latter half of *Phase 3* experienced further turbulent stormy weather, again evidenced in the NAO. Intermittent increases in abundance of the cyanobacteria, *Gloeotrichia spp.*, points to the occurrence of discrete periods of climatic warming, suggesting *Phase 3* was milder and further emphasises the turbulent nature of the climate during this final phase (Kosten et al, 2012).

The ecohydrology of *Phase 3* can be characterised by clear freshwater oligotrophic lake conditions with a hydrochemistry that ranges from circumneutral-slightly acidic towards one that is more circumneutral-acidic over time. The increase in emergent macrophyte

macrofossils such as *Equisetum fluviatile*, *Menyanthes trifoliata* and *Schoenoplectus lacustris* in this phase supports this. The rise in abundance of the acidophilous periphytic species *Tabellaria flocculosa*, which O'Driscoll et al (2012) found increased in abundance with declining alkalinities in upland peat rivers in NW Ireland, implies a shift towards more circumneutral-acidic conditions close to the surface following a more alkaline phase at around 10cm depth.

The 1901 and 1905 maps, which show a smaller lake and an adjacent wetland system, suggest that midway through *Phase 3*, the water level declined to a range more akin to the present day. Aerial photography from the last 60 years depicts some changes in the extent of the reedbed margins to the west and north of the lake, and it seems likely that it is this system that encroached during the early 1900s, and has since seen periods of growth and recession. At around 5cm to 7cm depth into the Core 1 and around 20cm depth in Core 2, there is a very slight increase in *Phragmites australis*, possibly evidence of a decline in water levels. From 7cm depth towards the surface, macrofossil abundance and richness decline and there is also a reduction in the dominant diatom species, *Achnanthes minutissima* and *Tabellaria flocculosa*. In contrast, *Fragilaria exigua*, which thrives under conditions of disturbance, displays an increase close to the surface. The fall in lake water levels displayed in the maps at the turn of the twentieth century are likely due to a combination of a climatic shift towards drier conditions and the lowering of the water table level relative to the ground surface due to significant aeolian sand deposition.

The final part of *Phase 3* is characterised by an increase in lake level from the early 1900s AD to the present day. This shift may in large part be due to the installation of the sluice in 1945 AD and also due to the increase in storm and flooding frequency across Ireland in the 1990s and 2000s AD. Indeed, deepening and straightening of the outflow channel via dredging in the 1960s due to upstream flooding in the Duvoge valley (Emer Magee pers comm. 23rd June 2012) indicates higher water levels. The increase in diatoms, the decrease in macrofossil abundance and increase in macrofossil richness at the surface of the sediment record, all indicate this recent rise in lake level is likely to have been highly variable (Bao et al, 2007).

5.5 Conclusions

The results in this chapter have revealed a lake and wetland system that has a complex past ecohydrology set in a dynamic coastal environment. The exploratory study provided the initial examination of the spatial and temporal variations and trends in the sedimentary sequences of the wider system upon which the more detailed lake core analysis was based. By analysing sedimentary data along with paleoecological indicators, the exploratory study determined that a more intensive sample-based paleoanalysis and chronological evaluation would be feasible within the lake; however the preservation of paleoecological indicators in the wetland meant no subsequent cores were extracted from the wetland. The complete core stratigraphies that were subsequently extracted from the lake and analysed in detail provided a more complete and comprehensive multiproxy

evaluation of the past ecohydrology of the lake and wetland ecosystem. Although these detailed lake core stratigraphies did not reveal a transition from estuarine to freshwater habitats, they did reveal two important climatic and geomorphological shifts that define the three key phases in the recent environmental and ecohydrological history of the lake and wetland system. Radiocarbon dating also provided a temporal context against which the past ecohydrological and environmental changes could be more effectively analysed.

The transitional shift that occurred 353 ± 37 yrsBP (prior to 1472-1630 (1549) AD) from a sandy environment to one that is dominated by peat, followed a period of drier than average conditions, replacing a habitat that was characterised by an aeolian sand-dominated environment with one that is characterised by a more stable peat-dominated environment. Most importantly, the hydrological system at this time would have been primarily riverine. In contrast, the second period from 1472-1630 (1549) AD to c.1800 AD was characterised by wetter than average climatic conditions that encouraged the development of flooded areas. With limited evidence of lacustrine features the site likely comprised an extensive wetland system with significant stands of *Phragmites australis* lining the Duvoge and Aberarachrin rivers. *Phragmites australis* dominates this period and marks the transitional shift that occurred between 237 yrsBP (1778 AD) and 180 yrsBP (1835 AD) from a primarily riverine habitat to a lacustrine environment. The most recent period is the most turbulent period in the history of the lake and is characterised by wet and stormy climatic conditions that promoted successive periods of aeolian sand deposition across the site and stimulated water level fluctuations that led to a proportion of the lake system developing into terrestrial wetland. These fluctuations are also apparent in the most recent sediment layers, suggesting the installation of the sluice has had a significant impact on the past ecohydrology of lake. In particular, it should be noted that the abundance of the rare macrophyte, *Najas flexilis*, has not increased in this most recent part of the sediment record.

The results in this chapter go a significant way towards reconstructing the environmental and ecological history of Sheskinmore Lough lake and wetland system since its formation. The ultimate goal of this chapter, however, was to gain a better understanding of how far removed the lake and its surrounding habitats are from their natural, pre-sluice state in order for conservation of Sheskinmore Lough SPA to be effective in the long-term. The results have shown that the lake currently exists in a period dominated by a combination of turbulent climatic conditions and anthropogenic impacts (primarily following the installation of the sluice) that have led to fluctuating water levels and an ecology that is unstable as a result. This chapter has therefore constructed the past context against which the contemporary reconstructions and future predictions within this thesis can be compared. Ultimately, this chapter has enhanced existing site knowledge and contributed to the wider historical understanding of coastal lakes and wetland systems necessary to inform conservation management from local to international level.

6 Modelling the future ecohydrology of Sheskinmore Lough

6.1 Introduction

The aim of this chapter is to model the hydrology of the Sheskinmore Lough system to establish the long-term behaviour of water levels and their effects on the ecology of the lake and wetland system in the future. The development, calibration and validation of the hydrological model for the Sheskinmore Lough lake and wetland system is described. The model is subsequently employed to assess the potential impacts of future climate change and hydrological management on the ecohydrology of Sheskinmore Lough.

Enhancing our understanding of the impacts of climate change and hydrological management on lake and wetland systems is vital for conservation success (Gilvear & Bradley, 2000; Hollis & Thompson, 1998; Thompson & Finlayson, 2000). This is not only because hydrological processes influence the edaphic and biological characteristics of freshwater systems, but also because many of the human-induced impacts on these habitats result from hydrological modifications to the system, or within its wider catchment (Mitsch and Gosselink, 2000; Thompson, 2012). Indeed, unforeseen impacts may arise from hydrological modifications driven by conservation-oriented management practices. For example, raising water levels or diversion drainage channels to establish and maintain conditions required by desirable wetland plant and animal species may have unwanted consequences (Thompson et al, 2004). Improvements in the ability to predict the potential knock-on impacts of such hydrological modifications before they are implemented are required in order to develop management schemes that will achieve their goals and avoid undesirable outcomes. There is therefore significant potential for studies using models that can accurately represent these hydrological modifications and predict the likely impacts of climate change on freshwater systems.

6.2 Hydrological modelling

6.2.1 Introduction to hydrological modelling

Models simplify complex systems, enabling modellers to better and more easily understand them, whilst also facilitating the development of new theories and the asking of 'What if?' questions (Singh, 2010). Hydrological models have been used extensively to simplify real hydrological systems, predict their likely responses and future behaviour, and provide improved understanding of their functioning and interaction with the wider environment (Brooks et al, 1991; Refsgaard et al, 1992; Jain et al, 1992; Thompson & Hollis, 1995; Lorup et al, 1998; Al-Khudhary et al, 1999; Karvonen et al, 1999; Christiaens & Feyen, 2001; Thompson et al, 2004; McMichael et al, 2006; Kingston et al, 2011; Thompson et al, 2015; House et al, 2016). Ward and Robinson (2000) noted that hydrologists have long regarded the development of models as key in the advancement of their understanding of drainage basin hydrological processes; however they also acknowledge that hydrological models do not ever fully represent real hydrological

systems, are only as good as the data quality available to them and the understanding of hydrological system behaviour.

Hydrological models are primarily developed for specific purposes rather than for general scientific investigation and, therefore, each model is developed to accommodate the hydrological data available for the particular system in question and to address the specific requirements of a given study. In addition, models vary depending on the spatial scale of the area they represent, from the micro scale (e.g. simulating the complexities of infiltration into soil over centimetres of land) to the macro scale (e.g. simulating a multinational drainage basin) (Singh, 2010). As a result there are numerous hydrological models in existence ranging from simple stochastic black box models, which link a certain input (e.g. rainfall) to an output (e.g. runoff) to more complex physically based models that simulate the combination and interaction of real world hydrological processes. The advancement of hydrological modelling in the past five decades has led to a dramatic increase in our understanding of hydrological systems. Now considered a powerful technique of hydrological system investigation for both scientists and engineers, hydrological modelling forms the basis for modern hydrology (Singh, 2010).

Hydrological models can be classified according to system representation, hydrological process representation and spatial discretisation:

- 1. System representation:** this class comprises deterministic and stochastic hydrological models. Stochastic models are based on the principle that interactions between various catchment processes often cannot be expressed using simple mathematical equations (Yevjevich, 1987). Instead, stochastic models apply a random component to express the parts of reality that are uncertain. In deterministic models, two identical sets of inputs will produce the same output, while in stochastic models, identical inputs will generally result in different outputs if run under identical conditions (Abbott & Refsgaard, 1996). Deterministic models vary in complexity from simple cause-effect relationships describing the behaviour and interactions between various catchment processes using mathematical equations to more complex modelling systems (Yevjevich, 1987). They include: the Soil and Water Assessment Tool (SWAT); TOPMODEL; HBV; and MIKE SHE (Beven & Kirkby, 1979; Arnold et al, 1993; Lindström et al, 1997; Zhang & Lindström, 1997; Refsgaard & Storm, 1995; Butts et al, 2005).
- 2. Hydrological processes representation:** this class comprises empirical and conceptual models. Empirical models use empirical knowledge, statistical analysis or hydroinformatic data to represent the relationship between system outputs and inputs without any consideration of processes. They include: the unit hydrograph method; the Constrained Linear Systems (CLS) model; and the Artificial Neural Network (ANN) approach (Sherman, 1932). Empirical models are limited to the range of available data, offering little capability for prediction and assessment of changing catchment conditions (Zhang et al, 2008) or for application beyond the system for which they were developed. Conversely, conceptual models (the most common model type) capture dominant catchment dynamics while remaining

computationally efficient by using parameters that are not directly measurable and must be inferred or 'calibrated' from the observed data (Beven & Binley, 1992). They include: the O'Donnell model; HEC; and HYRRM (Shaw, 1994).

3. **Spatial discretisation:** this class comprises lumped, semi-distributed or distributed models. Lumped models ignore spatial variability and, using average process values, treat the entire catchments as one uniform hydrological entity (Refsgaard, 1996). They include: the Stanford Watershed Model; HYRRM; HBV; and NAM, the latter two of which can also be semi-distributed when applied to separate sub-catchments (Crawford & Linsley, 1966; Nielsen & Hansen, 1973; Blackie & Eles, 1985; Bergström, 1995; Singh, 1995). Conversely, semi-distributed or fully-distributed hydrological models take account of spatial variations in variables and parameters by segregating the watershed into a finite number of spatial units (Refsgaard, 1996). They include: THALES; SWAT; and MIKE SHE (Beven et al, 1987; Grayson et al, 1992; Refsgaard & Storm, 1995; Beven, 2001; Butts et al, 2005; Graham & Butts, 2005).

Modelling a catchment, or part thereof, firstly involves the decision of which model class to choose (Beven, 2001). Recently, the increased pressure on the world's natural resources and the growing global awareness of environmental issues has led to a shift in focus from discrete reservoir analysis, flood forecasting and control of point source pollution towards integrated management of catchments (Singh, 2010). In order to meet these challenges, there has been increasing emphasis on the development of physically-based distributed models (Ward and Robinson, 2000; Kingston et al, 2011; Thompson et al, 2015; House et al, 2016). Refsgaard (1996) emphasised that distributed physically-based hydrological models give a detailed and potentially more accurate description of catchment hydrological processes than other model types. Fully-distributed models also facilitate higher levels of understanding of catchment hydrology, including the spatial variability of features such as topography, soils, vegetation and meteorological conditions, by separately describing their distribution across the model domain via parameters and formula based on actual measured catchment properties (Grayson et al, 1993; Liang et al, 1994; Karvonen et al, 1999; Beldring et al, 2003; Das et al, 2008; Kingston et al, 2011; Thompson et al, 2015; House et al, 2016).

Their superior scope for application means distributed models have been widely used for a range of scientific and engineering purposes including undertaking environmental impact assessments for water resource management; investigating the influence of spatial variability in catchment climatic characteristics; and lake and wetland management (Refsgaard & Sorensen, 1994; Refsgaard & Sorensen, 1997; Thompson et al, 2004; Christensen & Lettenmaier, 2007; Zhang et al, 2008; Thompson et al, 2009) (Liang et al, 2004). As discussed above, examples of this type of model include MIKE SHE and the IHDM-model (Beven et al, 1987; Refsgaard & Storm, 1995; Butts et al, 2005). MIKE SHE, one of the most complex spatially-distributed deterministic hydrological models, is capable of thoroughly representing the complexity of rainfall-runoff processes within a catchment, whilst also incorporating the influence of land use on rainfall-runoff response. As such it is recognized as one of the most comprehensive hydrological models (Refsgaard

& Sørensen, 1997; Beven, 2002; Graham & Butts, 2005). The current study employs the MIKE SHE system to simulate river discharge and lake water levels in the Sheskinmore Lough catchment.

6.2.2 The MIKE SHE modelling system

MIKE SHE is a deterministic, fully-distributed and physically-based modelling system based on the *Système Hydrologique Européen* (SHE) model developed in the late 1970's and 1980's by an international consortium including DHI, IoH and SOGREAH (Abbott et al, 1986; Refsgaard et al, 2010). This consortium integrated the unsaturated and saturated zones together with overland flow into a complete dynamic system whilst maintaining interaction between the various components (Abbott et al, 1986; Jain et al, 1992; Xevi et al, 1997; Christiaens & Feyen, 2001; Thompson et al, 2004; Graham & Butts, 2005; McMichael et al, 2006; Refsgaard et al, 2010). Since the mid 1980s, it has been developed further by DHI Water & Environment (Graham & Butts, 2005; Refsgaard et al, 2010) to include pre- and post-processing interfaces and a suite of modules each allowing for the integration of a particular process of the water cycle into an integrated model.

MIKE SHE is coupled to MIKE 11, a one-dimensional hydraulic model, and the two are built and run within the graphical user interface, MIKE ZERO (Havnø et al, 1995; Thompson et al, 2004; Duranel, 2016). The model spatially distributes catchment characteristics and climate variables through an orthogonal grid network, comprising grid squares that extend in equal-sized columns both horizontally and vertically (Figure 6.1) (Graham and Butts, 2005). Figure 6.2 shows the available hydrological processes that can be simulated within MIKE SHE and specifies the numerical engines for computation of each hydrological process (Singh, 2010). A MIKE 11 river network when coupled with a MIKE SHE model is assumed to run along the boundaries of the MIKE SHE grid squares (Thompson et al, 2004; Xevi et al, 1997). The MIKE SHE Water Movement (WM) module has a modular structure comprising six process-oriented components that describe the major physical processes of the land phase of the hydrological cycle, and which include: precipitation; interception/evapotranspiration (ET); overland/channel flow (OC); unsaturated zone (UZ); saturated zone (SZ); snow melt (SM); and exchange between aquifers and rivers (EX) (DHI, 2005). The processes relevant to the present study are summarised in Table 6.1. Each of these processes can be represented at different levels of spatial distribution and complexity, according to the goals of the study, the availability of data and the modeller's choices (Butts et al, 2004).

MIKE SHE has been applied to a wide range of catchment sizes, from catchments and wetlands less than 10 km² in area (Thompson et al, 2004), to major international river basins spanning thousands of km² (Andersen et al, 2001; Andersen et al, 2002). The model has been widely used across the globe by scientists and consulting engineering companies to study a variety of water resource and environmental problems under diverse climatological and hydrological regimes (Refsgaard & Storm, 1995; Butts et al, 2005; Thompson et al, 2008; Zhang et al, 2008; Refsgaard et al, 2010). According to Graham and Butts (2005), MIKE SHE has been extensively used in the analysis and planning stages of

management schemes to address a wide range of ecohydrological problems in lakes, wetlands, river basins, surface and groundwater studies, agriculture, irrigation, soil studies, remote sensing, land use change and flood studies (Table 6.2). In addition, the MIKE SHE structure is very flexible, allowing the modeller to use components independently and customize it to local meet their individual data availabilities and modelling needs (DHI, 2005). Its modular structure also enables data exchange between components and the addition of new components (Oogathoo, 2006).

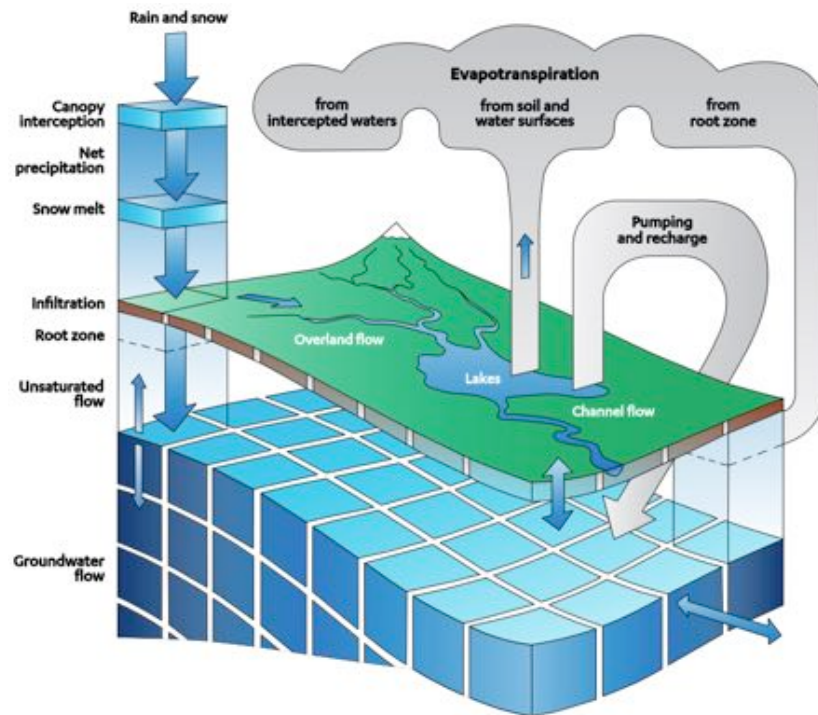


Figure 6.1 Hydrological processes simulated by MIKE SHE (DHI, 2005; p1).

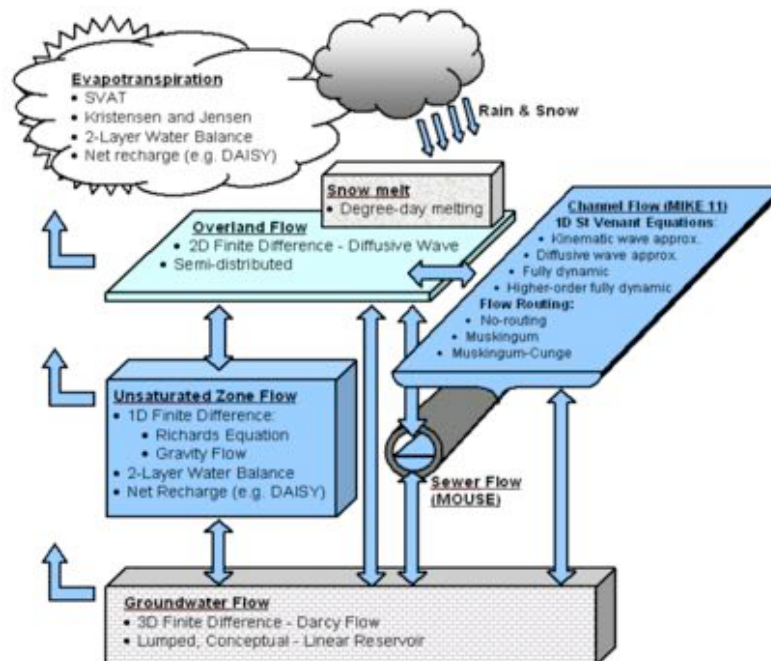


Figure 6.2 The numerical engines for each hydrological process available in MIKE SHE (Graham & Butts, 2005; p3).

Table 6.1 Summary of the MIKE SHE processes, data and methods relevant to the present study (Adapted from DHI, 2005).

Process	Data	Method
Precipitation	<ul style="list-style-type: none"> Precipitation rate (as a distribution and a value) Gridded Uniform, station-based or fully distributed 	<ul style="list-style-type: none"> Thiessen polygons or a similar distribution method applied to station-based precipitation data to spatially distribute it across the catchment An assembly of spatially discrete grid cells for the spatial representation of hydrological parameters and climate variables such as precipitation The precipitation data for a station located within each polygon is assigned to each grid cell within that polygon
Evapotranspiration (ET)	<ul style="list-style-type: none"> Meteorological Vegetation 	<ul style="list-style-type: none"> The ET model estimates actual ET in the root zone from potential evaporation, maximum root depth and leaf area index Also includes a simplified ET model for a two-layer unsaturated zone/ET model, which calculates the actual ET as well as the amount of water that recharges the saturated zone by dividing the unsaturated zone into two separate zones (a root zone, from which ET can be extracted, and a zone below the root zone, where ET does not occur)
Unsaturated Zone (UZ)	<ul style="list-style-type: none"> Soil moisture (replenished by precipitation and removed by ET and recharged to groundwater) 	<ul style="list-style-type: none"> The UZ is assumed heterogeneous, flow is estimated vertically The model calculates soil moisture and water table dynamics in the lower part of the soil profile by an iterative coupling process between the UZ and saturated zones Uses three methods to simulate UZ flow and moisture content: a) Richards Equation (using soil profiles that can have different soils at different depths); b) Gravity Flow (same as a) but with varying groundwater recharge data based on actual precipitation and ET observations); and (c) Two-Layer Water Balance methods (using a uniform soil for the entire depth, when the water table is shallow and soil moisture dynamics are not the focus of the study, and dividing the entire unsaturated zone into two layers that represent average conditions in the UZ, rather than detailed discretisation of the soil profile) Assumes that if sufficient water is available in the root zone then there is enough water available for ET Includes interception, ponding, infiltration, evapotranspiration and groundwater recharge
Saturated Zone (SZ)	<ul style="list-style-type: none"> Flow that interacts with overland flow, unsaturated flow, channel flow and evapo-transpiration 	<ul style="list-style-type: none"> Two methods for determining flow in the SZ: a) 3-Dimensional (3-D) finite difference method (spatial and temporal variation of hydraulic head using a 3-D Darcy mathematical equation and an iterative implicit finite difference technique to define the geological model, vertical numerical discretization and initial boundary conditions); and b) the linear reservoir method (the entire groundwater catchment is sub-divided into smaller sub-catchments and the water from these linear reservoirs is subsequently added to the river as lateral flow)
Overland Flow	<ul style="list-style-type: none"> Overland flow volume and direction 	<ul style="list-style-type: none"> Calculates using the diffusive wave approximation of the 2-D Saint-Venant equations (Finite Difference method) Or calculates using a semi-distributed approach based on the Manning equation (Simplified Routing method)
Channel Flow	<ul style="list-style-type: none"> Channel flow volume 	<ul style="list-style-type: none"> Computed using MIKE 11 and a finite difference scheme for unsteady flows in rivers and estuaries Can represent a wide range of hydraulic structures including weirs, sluices, gates, bridges and culverts
Aquifer to River Exchange	<ul style="list-style-type: none"> Exchange inflow or outflow 	<ul style="list-style-type: none"> Coupling between MIKE SHE and MIKE 11 via river links located on the edges of adjacent grid cells Three set up steps: a) establishment of a stand-alone MIKE 11 hydraulic model; b) establishment of a MIKE SHE model including overland flow, the UZ and the SZ; and c) coupling of MIKE SHE and MIKE 11 by defining branches where the two models can interact. Crucial for representation of river-aquifer interaction dynamics Enables simulation of inundation from MIKE 11 river model onto MIKE SHE grid squares

Table 6.2 Key areas of MIKE SHE application and their references (Adapted from Graham & Butts, 2005; and Singh, 2010).

Area of Application	References
Wetlands	Refsgaard & Sorensen (1994), Refsgaard et al (1994, 1998), Yan et al (1999), Jacobsen et al (1999), Thompson et al (2004), Thompson et al (2008), House et al (2016)
River basin management	Refsgaard et al (1992, 1998), Jain et al (1992), Refsgaard et al (1992), Refsgaard & Sørensen (1994), Sandholt et al (1999), Andersen et al (2001), Henriksen et al (2003), Vazquez (2003), Graham & Butts (2005)
Surface and groundwater modelling	Sørensen et al (1996), Refsgaard et al (1998), Kaiser-Hill (2001), Refsgaard et al (1998), Christiaens & Feyen (2001, 2002), Madsen & Kristensen (2002)
Soil studies	Storm et al (1987), Christiaens & Feyen (2001)
Agriculture	Thorsen et al (1998, 2001)
Irrigation	Carr et al (1993), Singh et al (1997, 1999), Jayatilaka et al (1998)
Remote sensing	Sandholt et al (1999, 2003), Andersen et al (2002), Butts et al (2004)
Land use change	Refsgaard & Sørensen (1994, 1997), Lørup et al (1998), Zhang et al (2008)
Flood studies	Butts et al (2005)

6.3 The MIKE SHE model of Sheskinmore Lough

6.3.1 Model development

The first stage of model development began by defining the complete or *whole* catchment area of Sheskinmore Lough, which is 22.19km² in area and encompasses the Duvoge and Abberachrin river catchments to the northeast and east of the lake, respectively, the dune system to the west and the downstream outflow catchment to the south (Figure 6.3). This was followed by determining the model grid cell size. Vásquez et al (2002) found little change in a number of model performance measures when MIKE SHE was applied over a range of grid cell sizes, while McMichael (2006) emphasised that the selection of model grid size should enable accurate representation of catchment attributes without placing excessive demands on computer run time.

Trial runs were undertaken on the *whole* catchment model using grid sizes of 100m², 70m², 40m² and 10m². Although data such as the catchment topography were described in more detail in the 10m² grid size, the computational time of the model run was considered excessive (approximately 72 hours), especially given the requirement to undertake multiple runs during calibration. Conversely, the 100m² grid size resulted in a reduced computational time (approximately 5 minutes); however the resolution of the processed data was considered too coarse to represent the catchment topography and other layers in the model. A grid size of 70m² was therefore applied as it represents an appropriate balance between detailed representation of the catchment layers and reasonable computation time: approximately 30 minutes.

In order to more accurately represent the immediate Sheskinmore Lough system, a smaller, more detailed catchment model with an area of 5.94km² and grid cell size of 40m² was subsequently employed. Like the *whole* catchment model, the grid cell size for this smaller *immediate* catchment model was chosen following trial runs of 50m², 40m², 30m² and 20m² grid sizes. 40m² was deemed optimum in terms of catchment detail and

computationally, with a run time of approximately 1 hour. Modelling of both the *whole* and *immediate* catchments was undertaken using the hydrological and meteorological data available for year one (18th June 2012 to 17th June 2013) of the two-year hydrological monitoring period from 18th June 2012 to 17th June 2014.

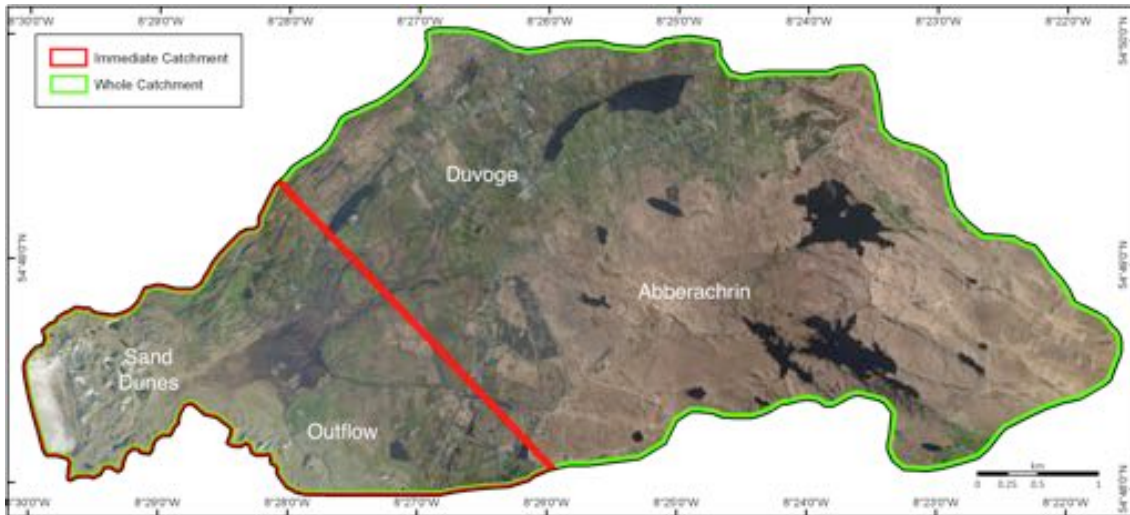


Figure 6.3 The *whole* and *immediate* catchments at Sheskinmore Lough.

The approach adopted was, firstly, to calibrate the *whole* catchment model with available discharge data from the two rivers for year one (18th June 2012 to 17th June 2013) of the hydrological monitoring period, and then validate the *whole* model with hydrological data collected during the subsequent year (18th June 2013 to 17th June 2014). This split sample approach to validation was considered appropriate given the data available, as alternative methods (e.g. using different sub-catchments) require datasets longer than two years in duration (Klemes, 1986; Xu, 1999; Henriksen et al, 2003; Singh, 2010). Prior to calibration, the accuracy of the Abberachrin and Duvoga river discharge data (calculated for the two-year hydrological monitoring period using flow duration curves derived from discrete flow gauging surveys in June 2012 and 2013 (see Section 3.3.4 for full methodology)), were assessed by comparing the Abberachrin and Duvoga river discharge data with the publically-available observed discharge data from the neighbouring Owenea catchment as this catchment has a similar geology, but is larger in area. The observed data were subsequently corrected via area weighting (Office of Public Works, 2014).

Secondly, the approach was to specify the modelled discharge result files from the *whole* calibrated model as boundary conditions in the smaller *immediate* catchment model, to ensure boundary conditions were representative of the complete catchment. The *immediate* catchment model was then calibrated (using the *whole* model final calibration values) and validated using the split sample approach over the same time periods. The effective calibration and validation simulation period for both models was only 9 months from 18th September 2012 to 17th June 2013 and from 18th September 2013 to 17th June 2014, respectively. This was because the first three months of each model simulation (18th June to 17th September 2012 and 18th June to 17th September 2013) were used as an initial model stabilisation period.

The MIKE SHE model development process for modelling the *whole* and *immediate* Sheskinmore Lough catchments was conducted in accordance with the structure of model components within the MIKE SHE software. In order for the model to run, each component of the model must be specified. The size and format of the map layers shown in Figure 6.3 are controlled within the model display, a component that sets the visual context for the remainder of the model development process. During simulation specification, different modules within the MIKE SHE model are selected. The Water Movement module, comprising Overland Flow (OL), Rivers and Lakes (OC), Unsaturated Flow (UZ), Evapotranspiration (ET) and Saturated Flow (SZ) modules, was selected to model the Sheskinmore Lough catchments. The simulation period was also specified in this module. Specifically, 24 hours was specified for the MIKE SHE maximum time step, which was deemed appropriate given the daily observations. A shorter MIKE 11 time step of 30 minutes was specified for computational stability reasons.

Precipitation and evapotranspiration data from the on-site weather station were used as input data for the MIKE SHE models. Comparison with regional datasets (Malin Head, Finner and Ballyhaise station data downloaded from Met Éireann (2015)), however, revealed that the observed precipitation data were underrepresented by approximately 33% (Figure 6.4). This is likely due to the impact of high wind speeds on the rain catch by the gauge of the weather station apparatus (Duchon & Essenberg, 2001). Daily precipitation data were therefore corrected by multiplying each daily data point by 1.33 to more accurately reflect regional precipitation patterns. Daily precipitation and daily evapotranspiration data for the calibration and validation periods were specified for both the *whole* and *immediate* catchment models. The precipitation and evapotranspiration data were uniformly distributed across both model catchment areas. Precipitation lapse rates were also applied to trial model runs; however the limited elevation range of the catchments meant the influence of these lapse rates was negligible and they were therefore omitted from the model setup.

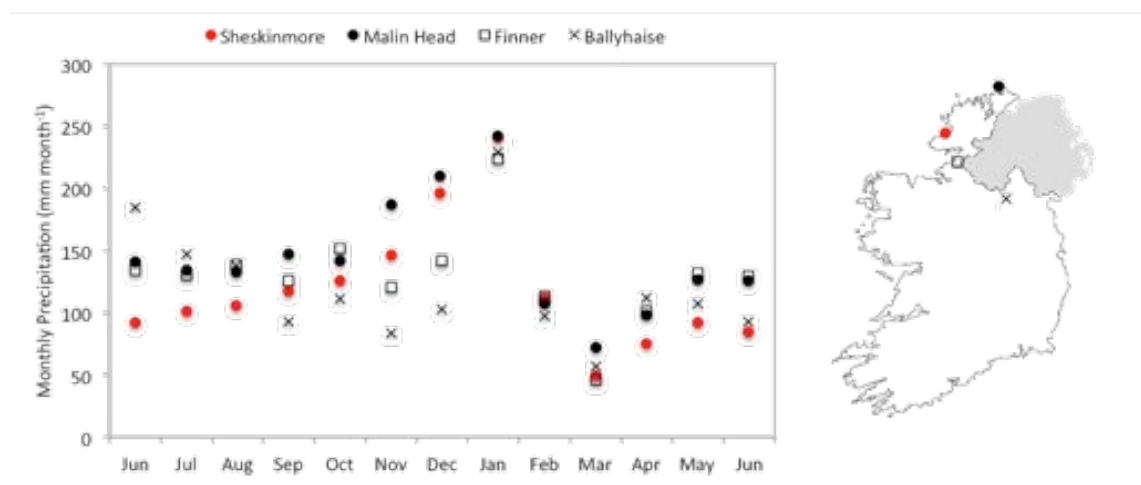


Figure 6.4 Comparison of monthly total precipitation (mm month⁻¹) data recorded at the Sheskinmore Lough and regional Malin Head, Finner and Ballyhaise weather stations (and their respective locations in Ireland) during the first year of the hydrological monitoring period (Met Éireann, 2015).

Topographical data of the Sheskinmore Lough catchment derived from contour and topographical survey data, which had a grid size of 1m^2 , were resampled via pre-processing to the 70m^2 and 40m^2 modelled grid sizes (Figure 6.5). The MIKE SHE resampling process uses the nearest neighbour assignment technique which determines the location of the closest cell centre in the input raster and assigns the value of that cell to the resulting cell on the output raster. The resampled topographic data was extracted to ASCII raster format and subsequently converted to the MIKE SHE dfs2 file format, which is used for gridded data. Figure 6.6 shows the hypsometric curves of the Sheskinmore catchment derived from the original and processed topographic data. The curves demonstrate that the processed topographic data represents similar topographic characteristic of the Sheskinmore catchment area to the original topographic data despite the reduction in resolution. The elevation of the *whole* Sheskinmore catchment lies between 0mOD and 93mOD, while the *immediate* catchment lies between 0mOD and 74mOD. In order to optimise simulation of lake water levels, inaccuracies in the processed topographic data from the *immediate* model were reduced via manual editing of the dfs2 file. This was achieved by comparing the elevation of each processed 40m^2 grid cell with the mean elevation of the same 40m^2 area within the original 1m^2 dataset. This was to ensure both the subtle variations in ground surface elevation within and around the lake area and the more sharp variations in elevation around the outflow were accurately represented.

The site habitat areas discussed in Chapter 4 were used to define the distribution of vegetation classes. Like the topographic data, the vegetation data were also resampled to the 70m^2 and 40m^2 grid sizes for the *whole* and *immediate* models, respectively (Figure 6.7). The original habitat area polygon shape files were resampled to a grid using ArcGIS, which assigns the most dominant land cover to that grid since it cannot interpolate to non-integer values. Table 6.3 demonstrates that the areas covered by the different vegetation types in both models exhibit little variation between the processed and original datasets. It also shows the Leaf area index (LAI) and root depth values that were defined for each vegetation class by taking estimates from the literature (Dittmer, 1959; Boggie & Knight, 1960; Persson & Baitulin, 1996; Steinke et al, 1996; Sorrel et al, 2000; Bradford & Acreman, 2003; Lalke-Porczyk & Donderski, 2004; Kalliokoski, 2011). In the absence of arable crops and deciduous woodland, LAI and root depth values were kept constant throughout the simulation period as monthly variation is minor and thus had negligible impact during trial runs.

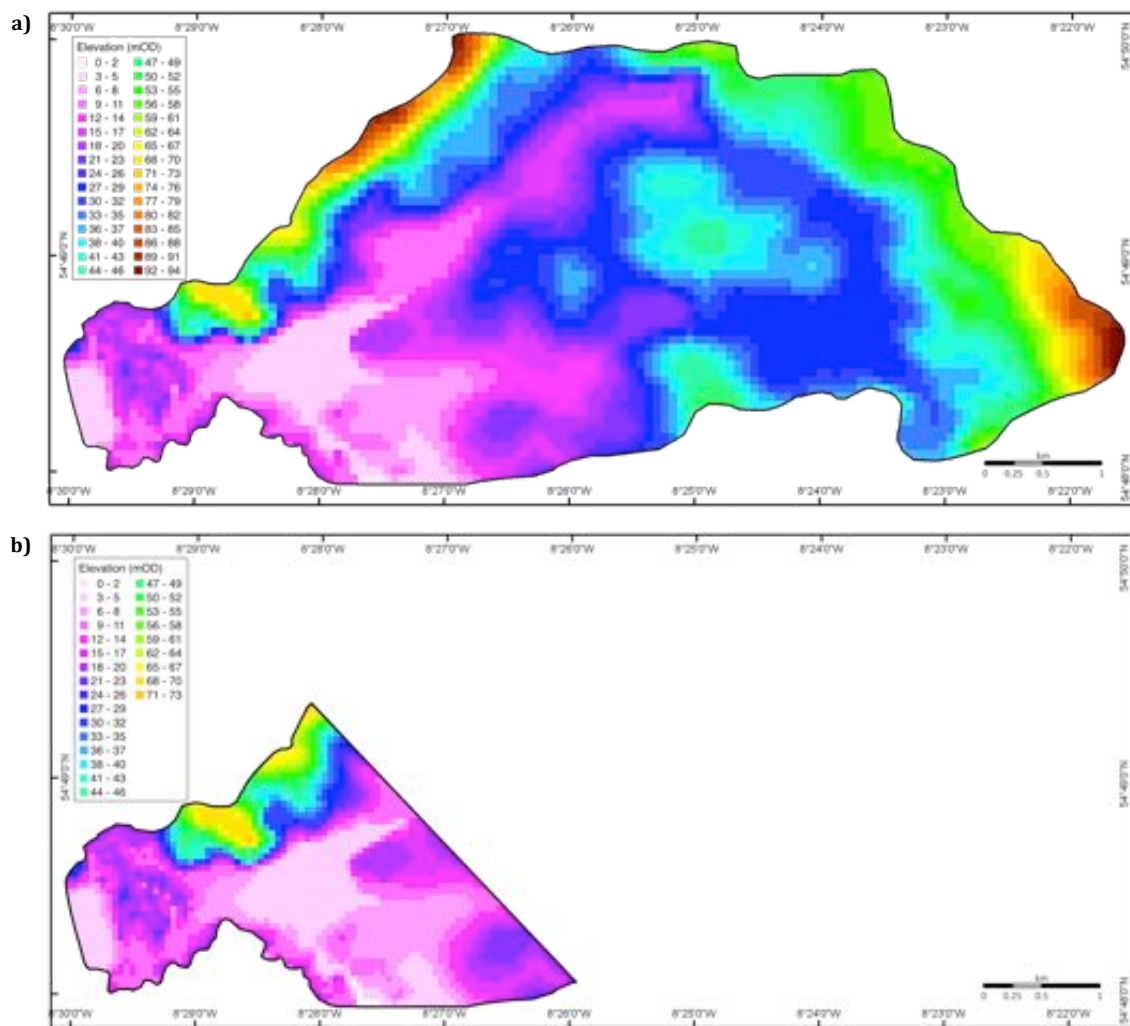


Figure 6.5 Processed topographic map (elevation mOD) of the a) *whole* model catchment (grid size 70m²) and b) *immediate* model catchment (grid size 40m²).

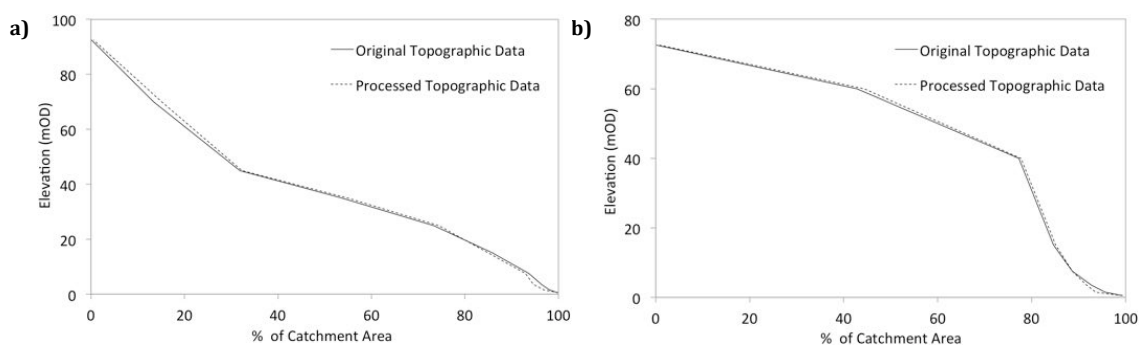


Figure 6.6 Hypsometric curves for the Sheskinmore topography in the a) *whole* and b) *immediate* catchments.

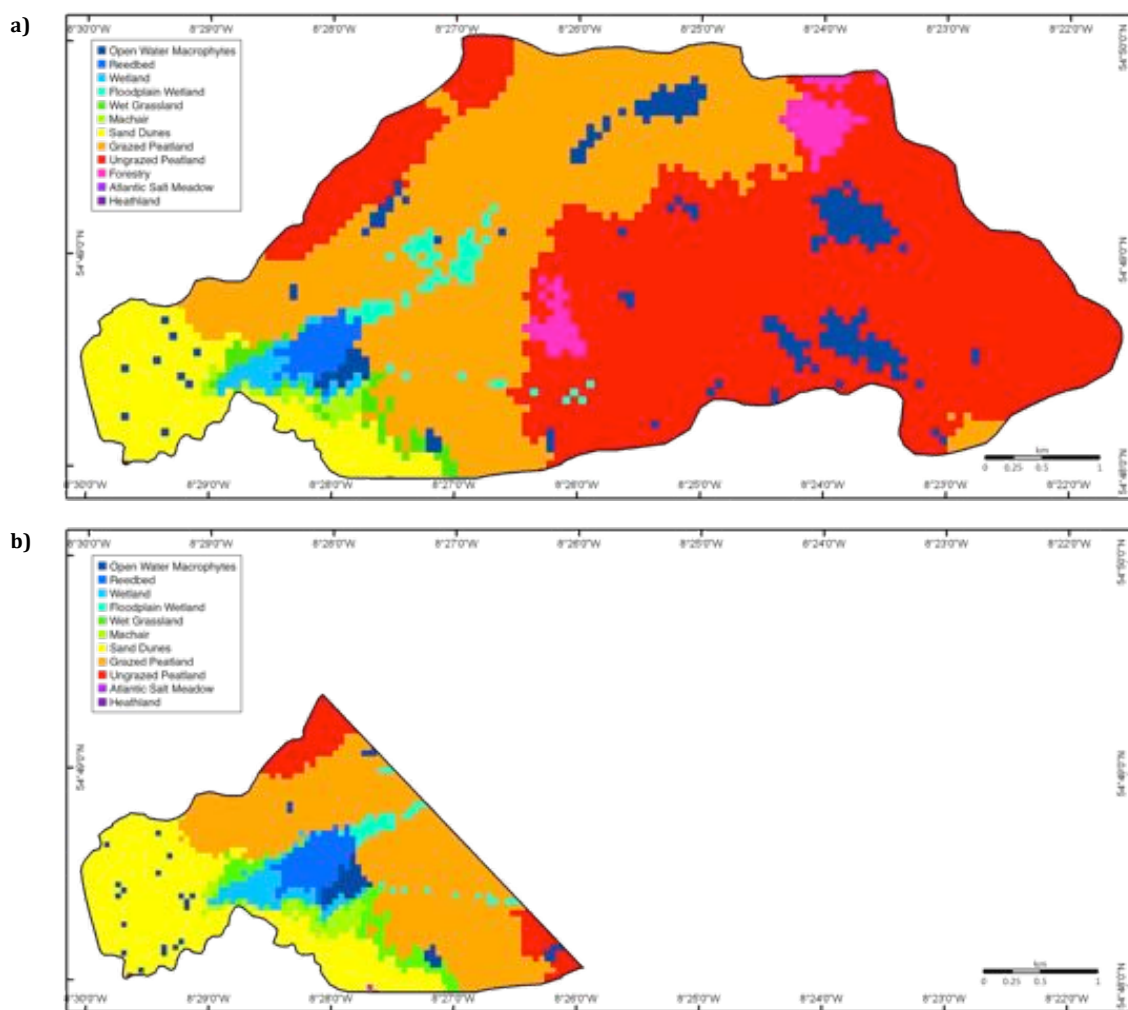


Figure 6.7 Processed vegetation areas in the a) *whole* model catchment (grid size 70m²) and b) *immediate* model catchment (grid size 40m²).

Table 6.3 Comparison of the original and processed land use vegetation type data, and their respective Leaf Area Index and root depth values for both the *whole* and *immediate* catchment models.

Vegetation Type	Whole Model		Immediate Model		Leaf Area Index	Root Depth (mm)
	Original Area km ² (%)	Processed Area km ² (%)	Original Area km ² (%)	Processed Area km ² (%)		
Open Water Macrophytes	1.14 (5.14)	1.12 (5.18)	0.17 (2.93)	0.17 (2.91)	2	500
Wetland	0.21 (1.96)	0.20 (0.93)	0.22 (3.66)	0.22 (3.63)	5	500
Reedbed	0.27 (1.20)	0.25 (1.18)	0.28 (4.64)	0.27 (4.56)	4.5	600
River Floodplain	0.31 (1.40)	0.30 (1.41)	0.11 (1.83)	0.11 (1.81)	4	500
Wet Grassland	0.27 (1.12)	0.25 (1.18)	0.27 (4.55)	0.27 (4.51)	2.5	450
Machair	0.16 (0.70)	0.15 (0.72)	0.16 (2.68)	0.16 (2.66)	3	300
Sand Dunes	1.92 (8.63)	1.87 (8.65)	1.92 (32.3)	1.88 (31.7)	2.75	350
Ungrazed Peatland	10.1 (45.5)	9.98 (46.3)	0.35 (5.84)	0.34 (5.74)	1.75	200
Grazed Peatland	7.34 (33.1)	6.97 (32.3)	2.46 (41.4)	2.51 (42.3)	3.5	150
Forestry	0.47 (2.10)	0.45 (2.09)	0 (0)	0 (0)	6	900
Atlantic Salt Meadow	0.02 (0.07)	0.02 (0.07)	0.01 (0.09)	0.01 (0.08)	3.25	200
Heathland	0.01 (0.02)	0.01 (0.02)	0.01 (0.09)	0.01 (0.08)	2.25	300
Total	22.2	21.6	5.94	5.93		

MIKE 11 HD models for each of the *whole* and *immediate* catchments were developed and linked to the respective MIKE SHE models. The coupling between MIKE 11 and MIKE SHE is made using river links, located at the edges of the closest MIKE SHE grid cells. A shape file featuring the main Abberachrin, Duvoge and outflow river channels was initially developed in ArcGIS. This was used to define the MIKE 11 river network within the River Network Editor, details of which can be viewed in Table 6.4. A balance was made during digitization between representation of the river network and the maximum number of H/Q points (points where water level and discharge are calculated) that can be specified within a MIKE 11 model (250 in this case). At the H-points water is transferred from the river network to adjacent MIKE SHE river links. Q-points were selected as near as possible to the two locations where flow gauging was carried out in the Abberachrin and Duvoge rivers, and also at the position of the sluice in the outflow.

Table 6.4 Details of the river network features for the MIKE 11 HD *whole* and *immediate* catchment models.

Network Feature	Abberachrin River	Duvoge River	Outflow River
No. of Branches	1	1	1
Total length of river (km)	7.92	5.23	1.18
No. of Cross Sections	70	45	20
Upstream Hydrodynamic Boundary Discharge (m^3s^{-1}) <i>whole</i>	0	0	N/A
Upstream Hydrodynamic Boundary Discharge (m^3s^{-1}) <i>immediate</i>	<i>whole</i> model MIKE 11 discharge time series results	<i>whole</i> model MIKE 11 discharge time series results	N/A
Elevation of Downstream Head Boundary (mOD)	N/A	N/A	0.75

The complete river network was formed in the Network Editor by digitizing a series of points along each river that were then joined to form branches. The branches of each river were subsequently linked via branch connectors. Figure 6.8 shows the sections of river network employed in the MIKE 11 HD model for the three rivers. The total length of the river network, along with the number of branches in the MIKE 11 HD model for the *whole* and *immediate* model catchments, are provided in Table 6.4. The branches were specified as coupled to MIKE SHE. A leakage coefficient of 0.01Ls^{-1} was applied throughout the river network owing to the sandy peat lining of the riverbeds (Langhoff et al, 2001).

Cross-sections were specified throughout the MIKE 11 river model to ensure the river elevations are representative of the surface topographic features (Figure 6.8). Due to limited availability of surveyed cross-section data for the three river channels (only three cross sections of each river channel were surveyed during the dGPS survey and all are located near the lake (see section 4.4.2 for full dGPS survey)), synthetic cross-sections throughout the river network were developed from channel widths measured within ArcGIS using aerial photography (Bing, 2013). The depths of the synthetic cross-sections were calculated via interpolation between the surveyed cross-sections and the synthetic cross-sections. The (surveyed and synthetic) cross-sections for the Abberachrin, Duvoge and outflow rivers were specified in the MIKE 11 HD model along the main channels of the three rivers (Table 6.4 and Figure 6.9). The depths of the cross-sections were specified as depth relative to the top of the stream bank, whose elevations were extracted from the

topographic grid produced during pre-processing within the MIKE SHE model. Figure 6.10 presents the longitudinal profiles of the three rivers. The Abberachrin and Duvoge profiles reveal a steep slope in the areas of higher elevation towards the eastern side of the catchment and relatively flat profiles near to where they confluence. In contrast, the outflow profile (shown in both graphs) is consistently gently sloping along its full course. A uniform Manning's Coefficient (n) for channel resistance of $0.02\text{m}^{1/3}\text{s}^{-1}$ was applied throughout the river network. This value is taken from the literature based on the type of channel, which is characteristically an earthen channel with few cobbles (Chow, 1959; Benson & Dalrymple, 1967; Aldridge & Garrett, 1973; Phillips & Tadayan, 2007).

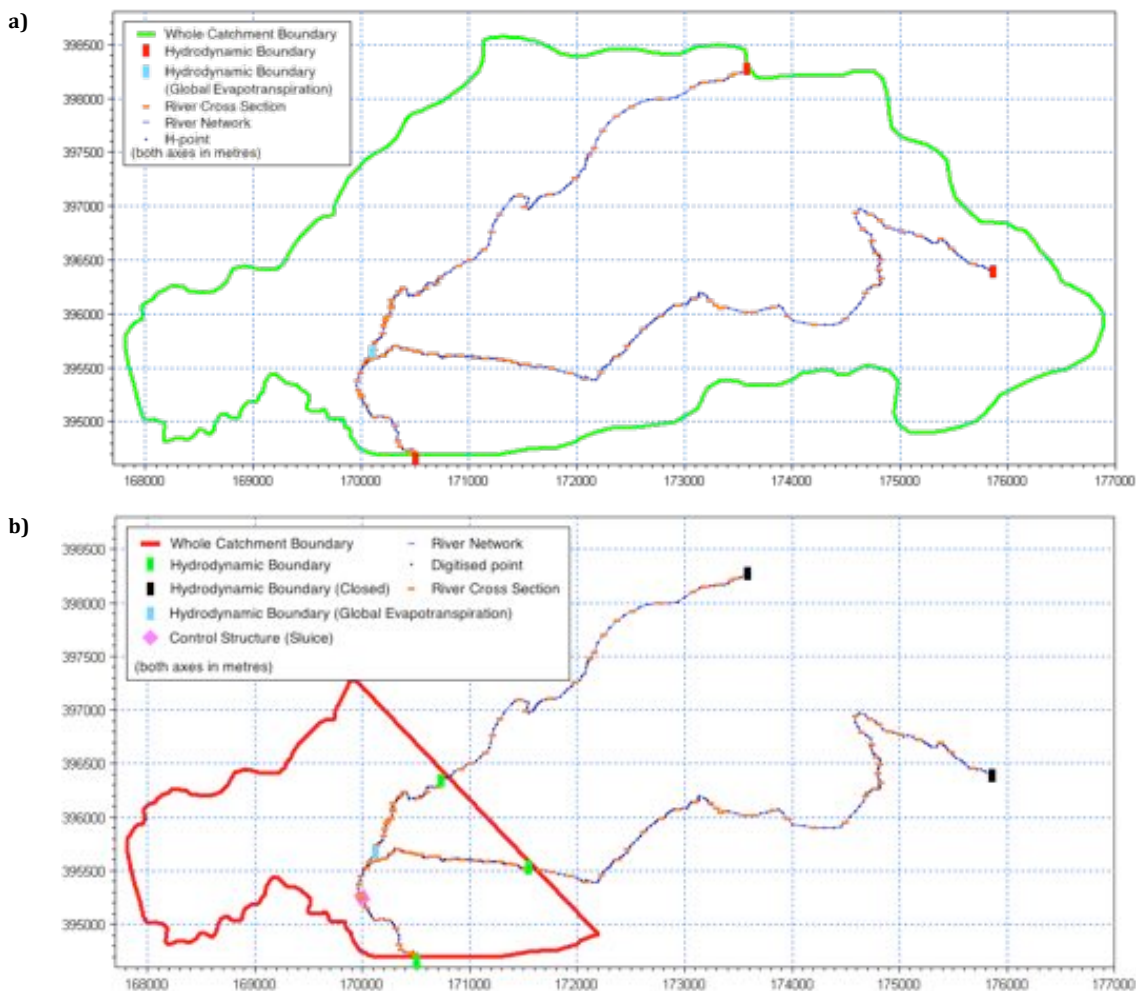


Figure 6.8 MIKE 11 models of the a) *whole* and b) *immediate* catchments.

Zero flow hydrodynamic boundaries were applied to the upstream open ends (i.e. the sources, where observed channel discharge is $0\text{m}^3\text{s}^{-1}$) of the Abberachrin and Duvoge river channels in the *whole* catchment model (Table 6.4 and Figure 6.8). In contrast, the hydrodynamic boundaries at the upstream open ends of the Abberachrin and Duvoge river channels in the *immediate* catchment model were specified as discharge boundaries with inflows being specified as the simulated discharges at these points from the *whole* catchment model. The downstream open end of the outflow branch, which is associated with a water depth of approximately 1.5m, was assigned a fixed water-level boundary with a consistent elevation in both models (0.75mOD) to ensure water flowing within the

models was discharged with no drying out effect (Thompson et al, 2009). A global time varying hydrodynamic boundary of evapotranspiration was also applied in both models to represent losses of water from the lakes, ponds and river network via evapotranspiration. This was specified, firstly, using digital elevation models and observed lake level data in ArcGIS to define the surface of the open water (lake and pond) area of the site on a given day and, secondly, using the cross sections in MIKE 11 to define the equivalent for the rivers. Once combined for each day of the modelling period, the full dataset was used to adjust the observed evapotranspiration data to produce the global dataset.

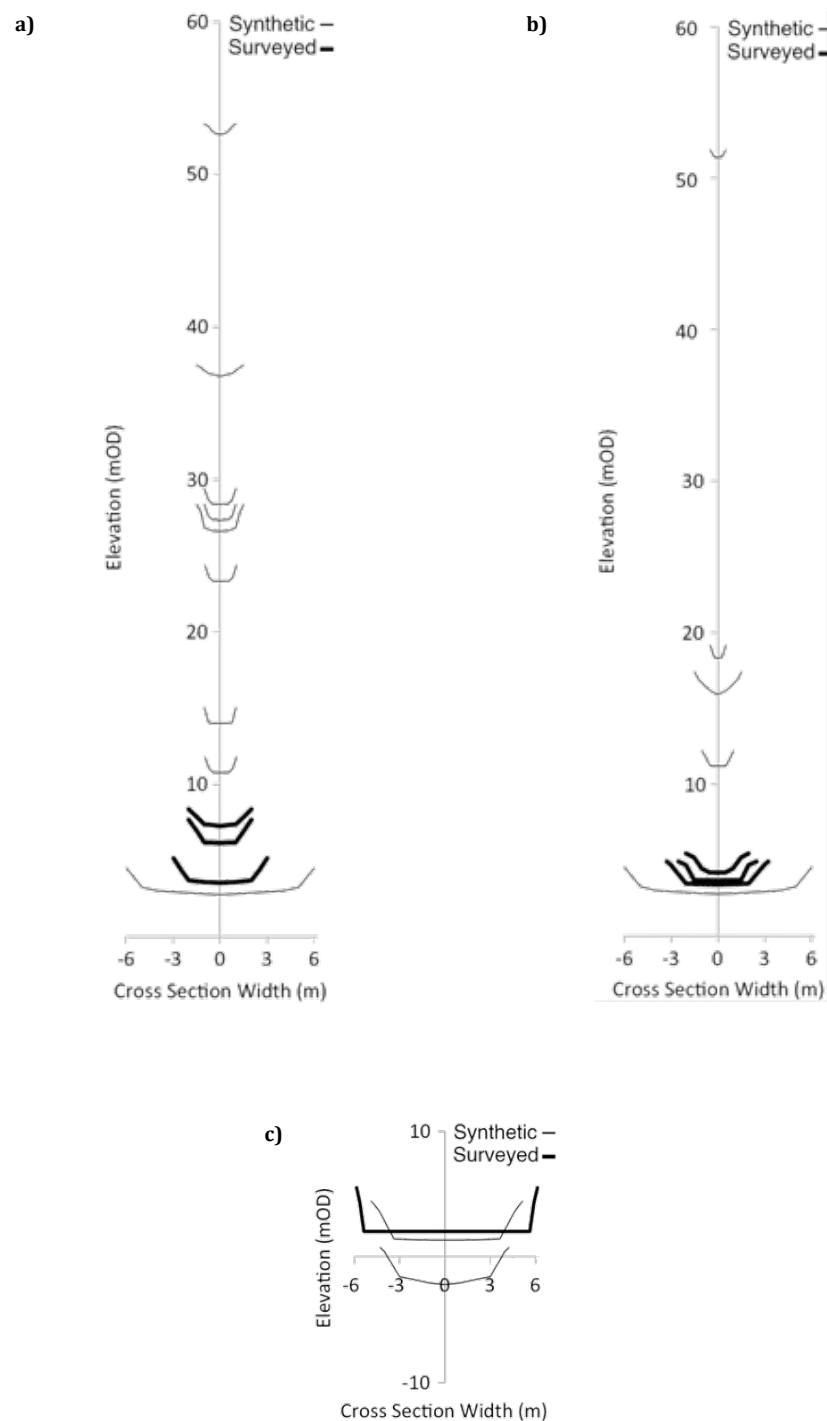


Figure 6.9 Cross sections at the sub-branch connections of the a) Abberachrin, b) Duvoge and c) Outflow rivers.

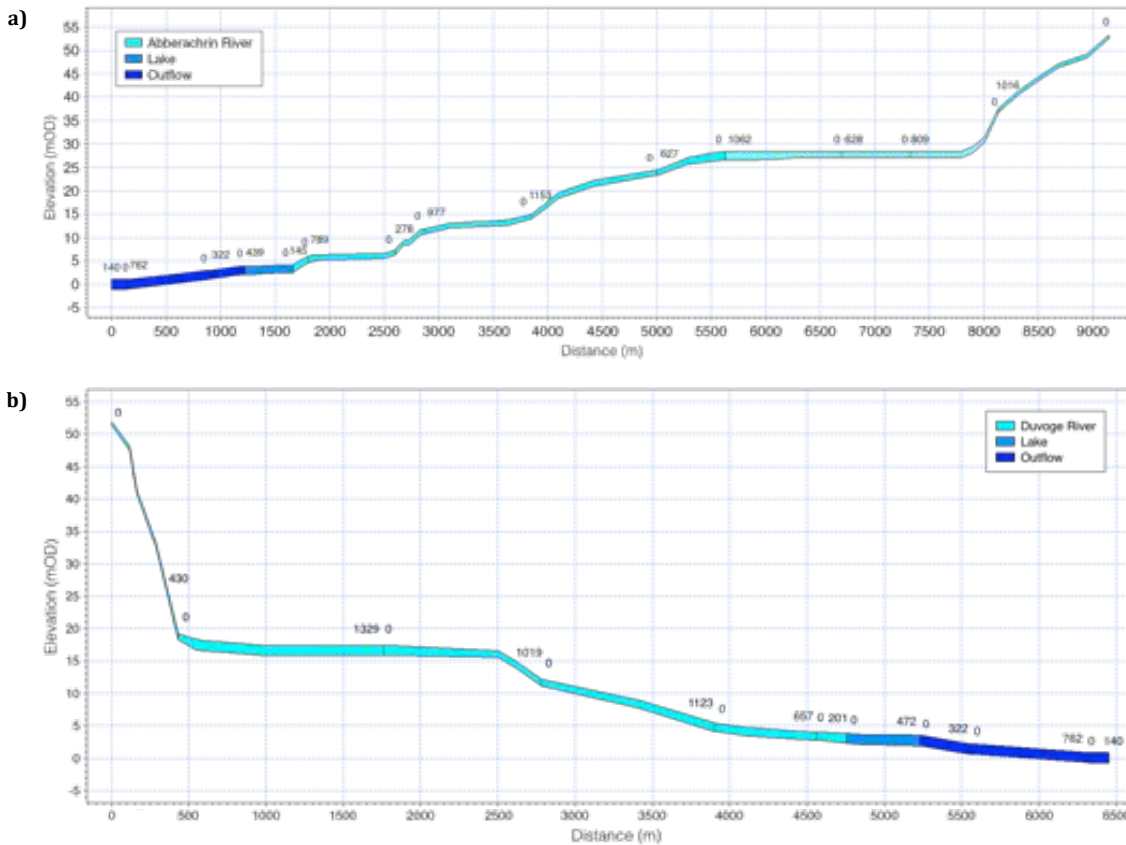


Figure 6.10 Longitudinal sections of the a) Abberachrin and Outflow and b) Duvoige and Outflow rivers labelled with sub-branch chainages and the relative position of the lake.

A ‘control structure’, in the form of a time varying ‘overflow gate’, was applied to the outflow river branch in the *immediate* catchment MIKE 11 model in order to represent the functioning effects of the sluice (Figure 6.8). An overflow gate structure was chosen as the most representative and computationally appropriate control structure type provided in MIKE 11. Table 6.5 displays the control structure configuration details, while the observed measurements of sluice elevation (i.e. openings (gate level: 2.02mOD) and closings (gate level: 2.02mOD)) (see section 4.4.2) are shown in Figure 6.11. This time series, recorded during the hydrological monitoring period from 18th June 2012 to 17th June 2014, determined the control structure levels throughout the modelling period.

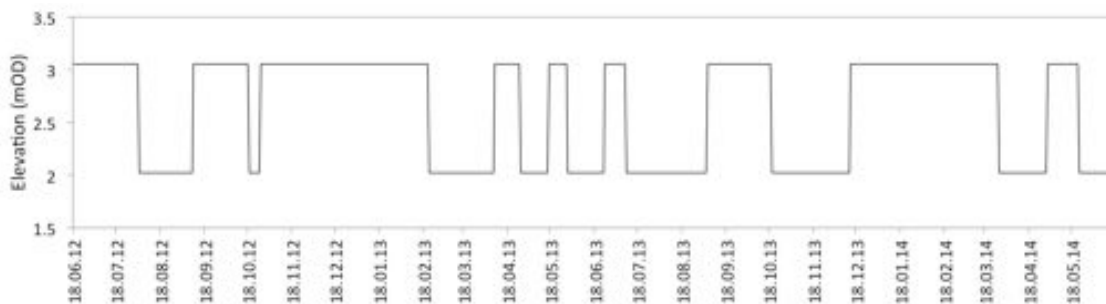


Figure 6.11 Observed time series of sluice operation (opening and closing) during the hydrological monitoring period from 18th June 2012 to 17th June 2014.

Table 6.5 Control structure (sluice) configuration details in the *immediate* catchment MIKE 11 model.

Control Structure Feature	Configuration
Control Type	Time
Target Type	Gate Level
Gate Type	Overflow
Gate Width	6m
Sill (Bed) Level	2.02mOD
Branch	Outflow
Chainage	299m

Surface water flow for the Sheskinmore Lough catchment was estimated using the finite difference method. The initial water depth on the ground surface was specified as zero. Given the nature of the ground surface across the catchment is not smooth, a detention storage depth of 0.01mm was specified as the minimum ponded water depth that has to be reached before overland flow occurs (Dingman, 1994; Duranel, 2016). A uniform Manning M value of $10\text{m}^{1/3}\text{s}^{-1}$ was specified, which was later modified during the calibration process (Chow, 1959; Benson & Dalrymple, 1967; Aldridge & Garrett, 1973; Phillips & Tadayon, 2007).

As discussed in section 2.3, the substrate in the Sheskinmore Lough catchment is characterised by a sedimentary drape of calcareous dune sand over acidic peatland, which in turn overlay metamorphosed granitic bedrock. The dominant substrate type in the catchment is peat, which is compact and relatively impermeable (hydraulic conductivity: $1 \times 10^{-3}\text{ms}^{-1}$ (Bear, 1972)). In contrast, the drape of calcareous dune sand is confined to the southwest side of the catchment, is less compact and has a higher permeability (saturated hydraulic conductivity: 0.01ms^{-1} (Bear, 1972)). The unsaturated zone was therefore represented by two spatially distributed soil types: sand and peat (Figure 6.12). These were defined in ArcGIS by delineating the sand dune system from aerial photography (Bing, 2013), while the remainder of the model area was specified as peat. Table 6.6 shows that there is little difference in the extent of these two different soil types between the original and processed data. The initial hydraulic parameters applied in the Unsaturated Zone in Table 6.6 were derived from the literature and subsequently modified during calibration (Mualem, 1976; Letts et al, 2000).

In the absence of detailed geological information, a single uniform 3m thick peat layer was specified in the saturated zone module for both the *whole* and *immediate* catchments. A sand layer of variable thickness was defined from the DEM by converting elevation into depth below the ground surface in the areas of the catchment where sand was observed during the field surveys (Figure 6.13). The resampled sand layer data was extracted to ASCII raster format and subsequently converted to the MIKE SHE dfs2 file format. Figure 6.14 shows the hypsometric curves of the Sheskinmore catchment derived from the original and processed saturated zone sand depth data. The curves demonstrate that the processed depth data represents similar characteristics to the original depth data despite the reduction in resolution. The depth below ground surface of the *whole* Sheskinmore catchment lies between 0mOD and 93mOD, while the *immediate* catchment lies between 0mOD and 74mOD. The initial parameter values in Table 6.8 were taken from the

literature and were later used in model calibration (Mualem, 1976; Letts et al, 2000). Saturated Zone flow was estimated by employing the 3D finite difference method. An outer boundary defined by the catchment area in both the *whole* and *immediate* models were specified with a zero flux (no-flow) boundary condition.

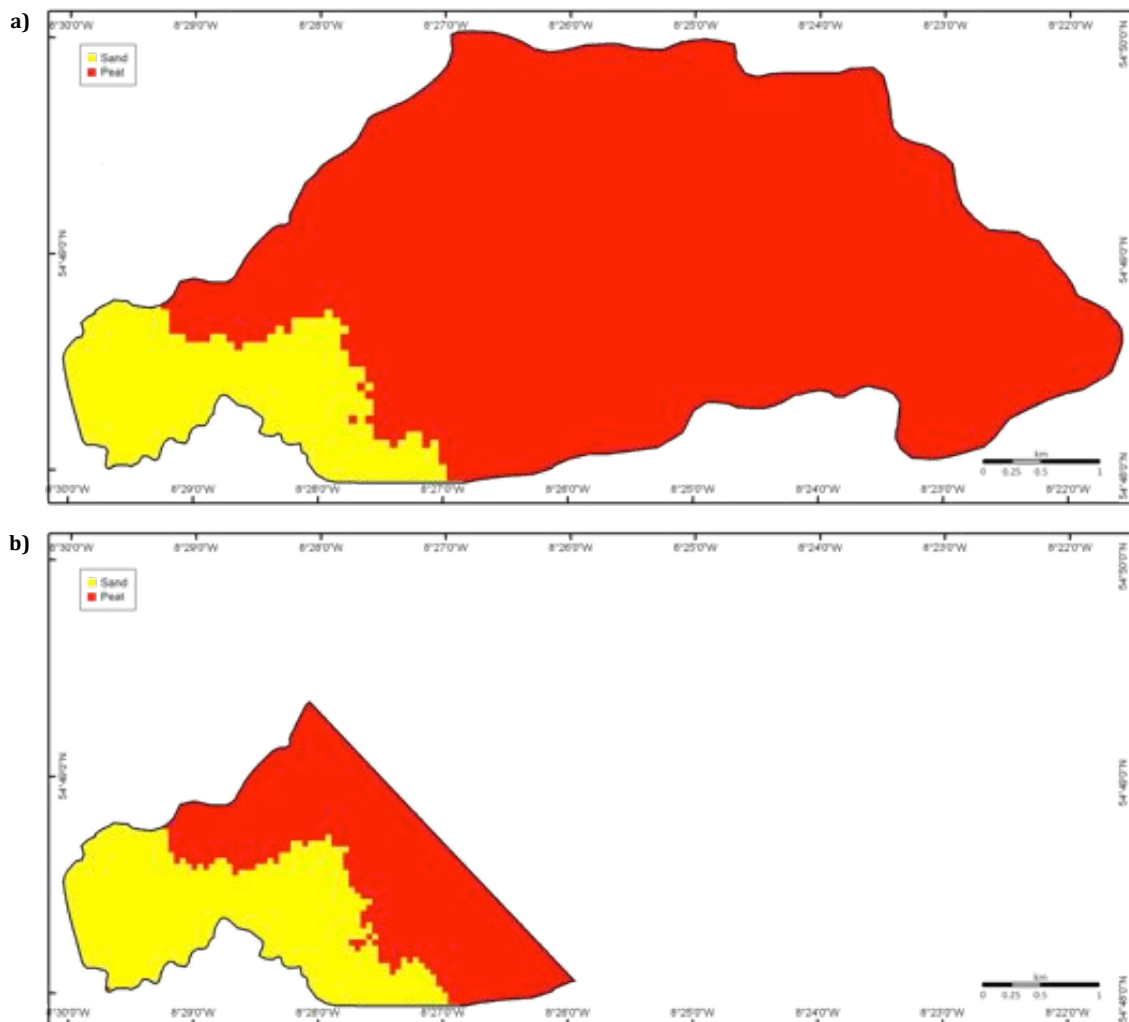


Figure 6.12 Processed Unsaturated Zone soil types in the a) *whole* model catchment (grid size 70m²) and b) *immediate* model catchment (grid size 40m²).

Table 6.6 Comparison of the original and processed Unsaturated Zone soil areas for both the *whole* and *immediate* models.

Soil Type	<i>Whole Model</i>		<i>Immediate Model</i>	
	Original Area km ² (%)	Processed Area km ² (%)	Original Area km ² (%)	Processed Area km ² (%)
Sand	3.03 (13.6)	2.98 (13.4)	2.95 (49.7)	2.95 (49.7)
Peat	19.2 (86.4)	19.0 (86.6)	2.99 (50.3)	2.98 (50.3)
Total	22.2	22.0	5.94	5.93

Table 6.7 Hydraulic parameters and the initial values specified for the Unsaturated Zone soil types.

Hydraulic Parameter	Peat	Sand
Water content at saturation	0.4	0.4
Water content at field capacity	0.3	0.3
Water content at wilting point	0.2	0.2
Saturated hydraulic conductivity (ms ⁻¹)	1 x 10 ⁻³	0.01

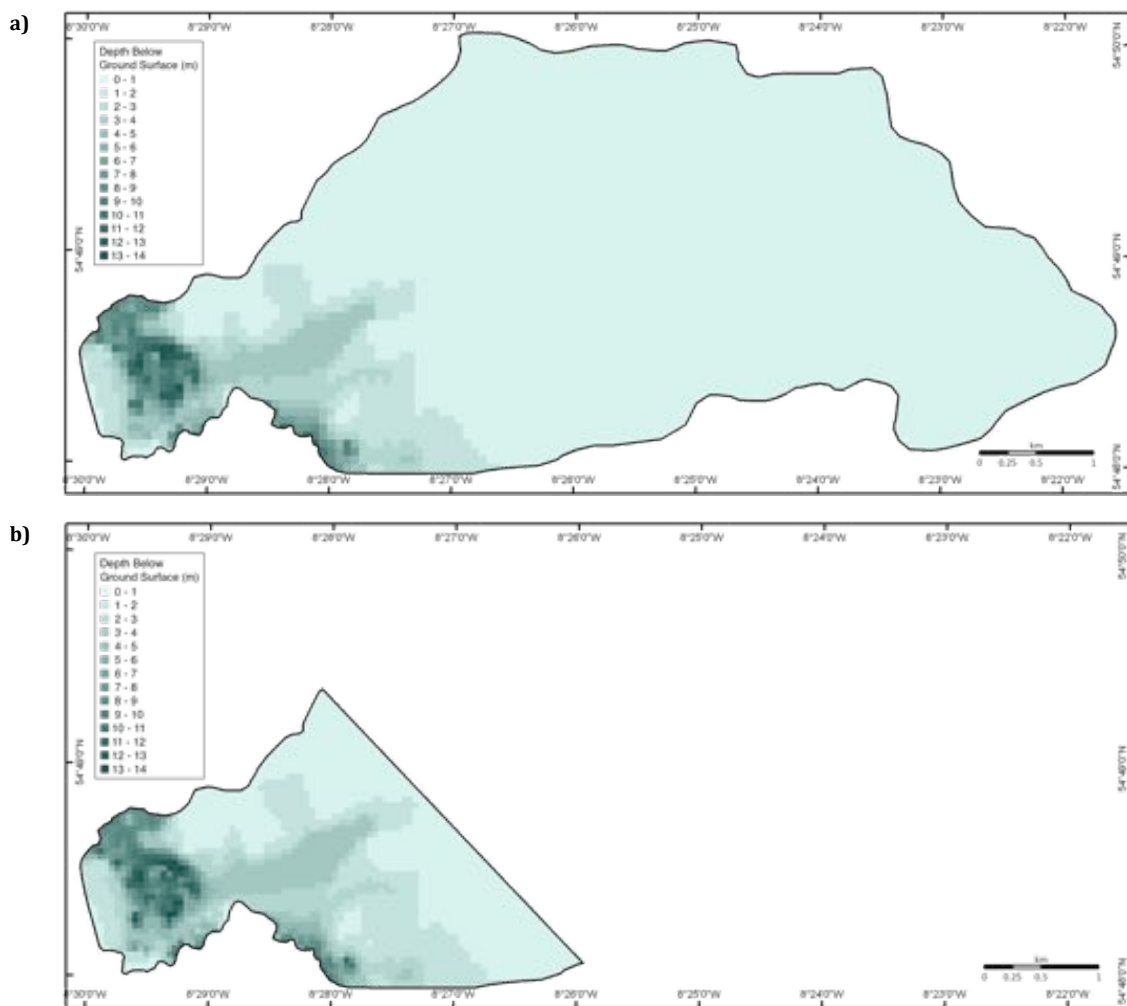


Figure 6.13 Processed Saturated Zone sand layer in the a) *whole* model catchment (grid size 70m²) and b) *immediate* model catchment (grid size 40m²).

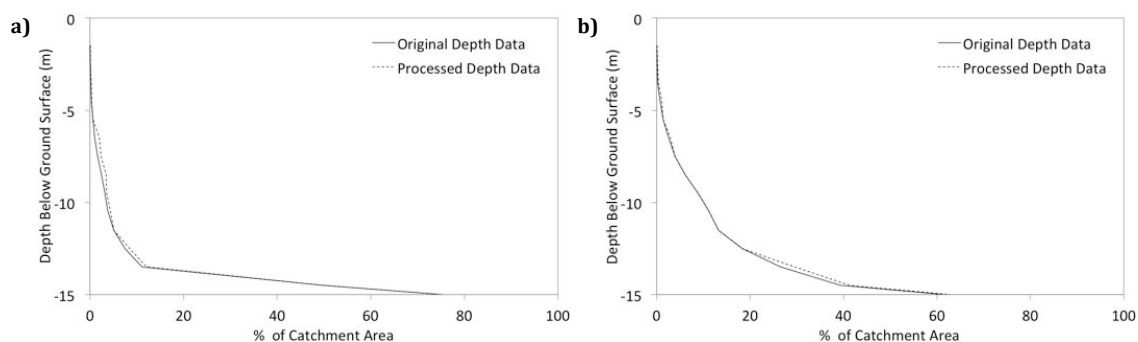


Figure 6.14 Hypsometric curves for the Sheskinmore saturated zone sand layer depth in the a) *whole* and b) *immediate* catchments.

Table 6.8 Hydraulic parameters specified for the Saturated Zone soil types.

Hydraulic Parameter	Peat	Sand
Horizontal hydraulic conductivity	$1 \times 10^{-7} \text{ ms}^{-1}$	0.0001
Vertical hydraulic conductivity	$1 \times 10^{-7} \text{ ms}^{-1}$	0.0001
Specific yield	0.2	0.1
Storage coefficient	0.0001	0.001

6.3.2 Model calibration and validation

Refsgaard & Storm (1995) suggested that the number of parameters subjected to adjustment during calibration within a distributed hydrological model such as MIKE SHE should be as small as possible. Indeed, studies modelling UK wetlands have tended to use a limited array of calibration parameters for MIKE SHE and MIKE 11 models (Al-Khudhairy et al, 1999; Thompson et al, 2004; Graham & Butts, 2005; Thompson et al, 2009). In these studies, parameter adjustments are primarily limited to hydraulic conductivity in the saturated zone, the Manning's roughness coefficient for overland as well as channel flow, the channel leakage coefficient and the drainage time constant used in the representation of sub-grid scale surface drainage. This thesis subjected a similar array of parameters in the *whole* catchment model to calibration. These parameters are listed in Table 6.9. As described above, initial values were obtained from the literature (Chow, 1959; Benson & Dalrymple, 1967; Aldridge & Garrett, 1973). The *immediate* catchment model inherited the same calibrated parameters from the *whole* model.

As described in Section 6.3.1, initially the *whole* Sheskinmore catchment model was calibrated against the available observed discharge data from the Abberachrin and Duvoge rivers for the first year of the hydrological monitoring period from 18th June 2012 to 17th June 2013. Calibration was carried out using a manual iterative procedure and the performance of each model run was assessed based on a graphical comparison of observed and simulated river discharge data and widely used statistical measures of model performance: the correlation coefficient (R) (Weglarczyk, 1998; Yang et al, 2002) and the Nash–Sutcliffe coefficient (R²) (Nash & Sutcliffe, 1970; Garrick et al, 1978; Xiong & Gou, 1999; Andersen et al, 2001; Thompson et al, 2004).

Similar to previous modelling studies (Jain et al, 1992; Thompson et al, 2004), the *whole* catchment model was most sensitive to changes in hydraulic conductivity (both vertical and horizontal). Initial calibration model runs therefore involved modifying the hydraulic conductivity, while subsequent runs focused on fine-tuning model results through further modification of the other parameter values. The initial and final calibration parameter values are shown in Table 6.9. Once calibrated, the modelled discharge result files from the *whole* model were then specified as the upstream boundary conditions for the two inflowing streams to the smaller *immediate* catchment model for which the final calibration parameter values from the *whole* model were specified. Calibration of the *immediate* model focused purely on graphical and statistical comparison of the observed and simulated MIKE SHE overland water level elevation in the area of the lake. Calibration runs of this model therefore featured adjustments to the open and closed elevations of the sluice structure. This was to ensure observed conditions were mirrored, ensuring the simulated overland water level elevation data followed the same pattern of response.

Table 6.9 Initial and final calibrated parameter values in the MIKE SHE and MIKE 11 models.

Model	Parameter	Initial Value	Final Calibration Value
MIKE SHE	Saturated Zone – Vertical Hydraulic Conductivity (Sand)	$1 \times 10^{-6} \text{ ms}^{-1}$	$1 \times 10^{-5} \text{ ms}^{-1}$
	Saturated Zone – Vertical Hydraulic Conductivity (Peat)	$1 \times 10^{-8} \text{ ms}^{-1}$	$1 \times 10^{-8} \text{ ms}^{-1}$
	Saturated Zone - Horizontal Hydraulic Conductivity (Sand)	$1 \times 10^{-6} \text{ ms}^{-1}$	$1 \times 10^{-5} \text{ ms}^{-1}$
	Saturated Zone - Horizontal Hydraulic Conductivity (Peat)	$1 \times 10^{-8} \text{ ms}^{-1}$	$1 \times 10^{-8} \text{ ms}^{-1}$
	Overland Flow Resistance (Manning's M)	$10 \text{ m}^{1/3} \text{ s}^{-1}$	$20 \text{ m}^{1/3} \text{ s}^{-1}$
	Unsaturated Zone Infiltration Rate (Sand)	$1 \times 10^{-7} \text{ ms}^{-1}$	$1 \times 10^{-4} \text{ ms}^{-1}$
	Unsaturated Zone Infiltration Rate (Peat)	$1 \times 10^{-8} \text{ ms}^{-1}$	$1 \times 10^{-10} \text{ ms}^{-1}$
MIKE 11	Bed Resistance of the Stream Channel (Manning's n)	$0.02 \text{ m}^{1/3} \text{ s}^{-1}$	$0.03 \text{ m}^{1/3} \text{ s}^{-1}$

As mentioned above, validation of the *whole* and *immediate* catchment models was carried out using a split sample approach (Klemes, 1986; Xu, 1999; Henriksen et al, 2003). The final parameter values from model calibration (Table 6.9) remained unchanged during validation. The calibrated models were run over the subsequent year period from 18th June 2013 to 17th June 2014. Validation of the models was based on comparisons of observed and simulated Abberachrin and Duvoge river discharges in the case of the *whole* model and lake water level in the case of the *immediate* model. These comparisons were supported by the same statistical measures of model performance employed in model calibration.

Figure 6.15 shows the observed and simulated Abberachrin and Duvoge daily and monthly river discharge for the *whole* catchment model over the first year of the hydrological monitoring survey period. These results demonstrate the model is generally successful at reproducing the observed daily and monthly discharge despite the flashy nature of the catchment's response to precipitation. After the warm up period of the first three months, the model achieves good sequencing of peak discharge, although the magnitudes of the largest peaks are underestimated in both rivers, especially the Abberachrin. The average of the 10 largest peak daily discharges in each river were simulated as 32% lower than observed in the Abberachrin ($1.94 \text{ m}^3 \text{ s}^{-1}$ (observed); $1.31 \text{ m}^3 \text{ s}^{-1}$ (simulated)) and 12% lower than observed in the Duvoge ($1.23 \text{ m}^3 \text{ s}^{-1}$ (observed); $1.08 \text{ m}^3 \text{ s}^{-1}$ (simulated)). The troughs in daily discharge in the Abberachrin are also underestimated by 10%. In April and May 2013 the model overestimates the daily discharge, especially the magnitude of the peak on 18th April 2013 in the Duvoge ($0.25 \text{ m}^3 \text{ s}^{-1}$ (observed); $0.33 \text{ m}^3 \text{ s}^{-1}$ (daily simulated)). The model also overestimates the monthly discharge, during these latter months in 2013 (April: $0.25 \text{ m}^3 \text{ s}^{-1}$ (observed), $0.40 \text{ m}^3 \text{ s}^{-1}$ (simulated); May: $0.54 \text{ m}^3 \text{ s}^{-1}$ (observed), $0.48 \text{ m}^3 \text{ s}^{-1}$ (simulated)).

The statistical measures of model performance confirm the ability of the *whole* catchment model to simulate daily and monthly discharge in the Abberachrin and Duvoge rivers during the calibration period (Figure 6.15 and Table 6.10). According to Henriksen et al (2008) classification, statistical analyses reveal the model is 'Very Good' ($R = 0.964$; $R^2 =$

0.676) at simulating daily Abberachrin river discharge, and 'Excellent' ($R = 0.997$; $R^2 = 0.885$) at simulating daily Duvoge river discharge using the Henriksen et al (2008) classification scheme (Figure 6.15 and Table 6.10). Similarly, the model is 'Very Good' at simulating monthly discharge in the Abberachrin ($R = 0.997$; $R^2 = 0.691$) and Duvoge ($R = 0.999$; $R^2 = 0.841$).

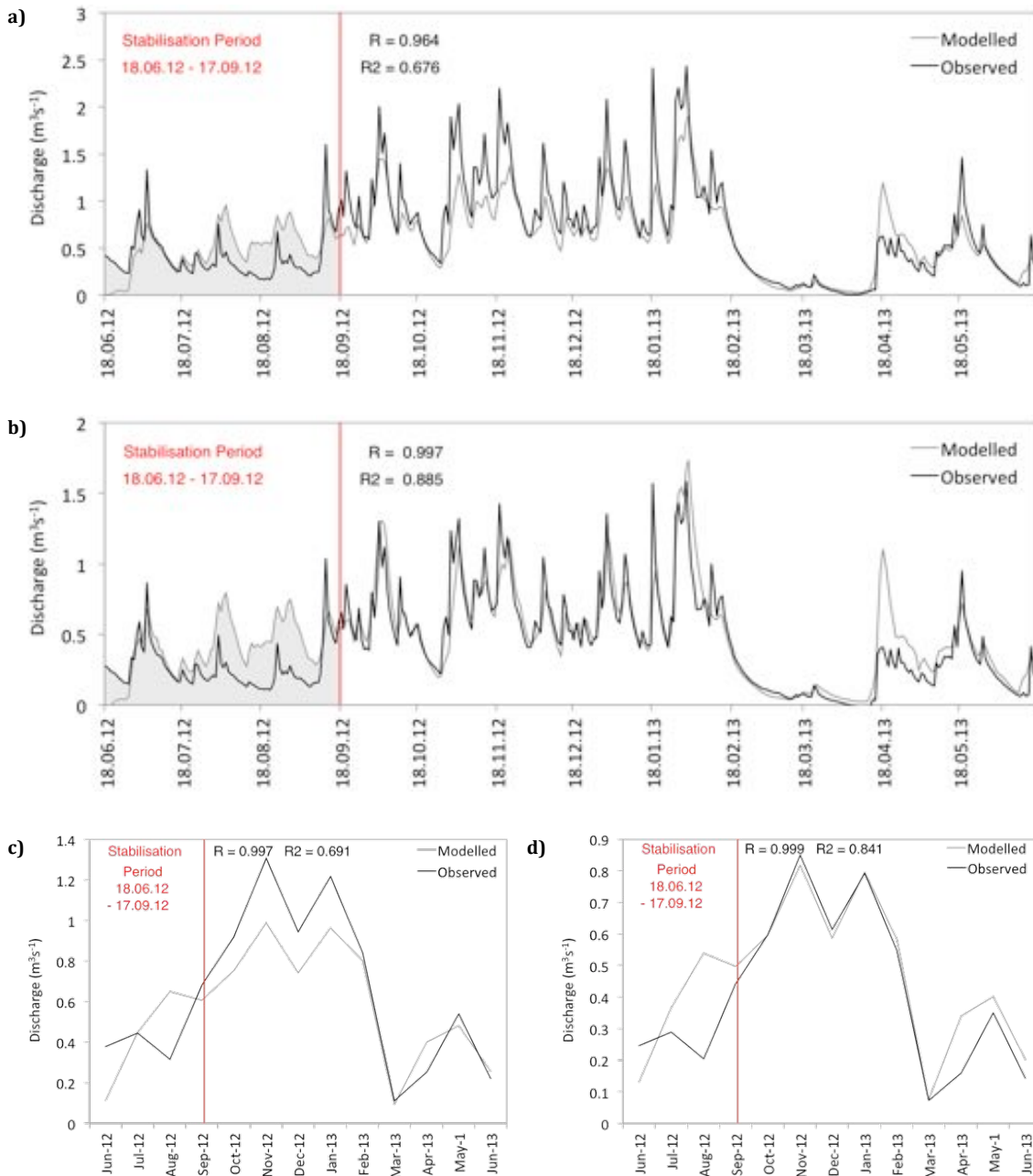


Figure 6.15 Comparison of daily and monthly observed and simulated discharge in the Abberachrin (a) and c)) and Duvoge (b) and d)) rivers in the *whole* model for the period of 18th June 2012 to 17th June 2013. Note the model stabilisation period from 18th June to 17th September 2012.

Table 6.10 Henriksen performance indicators for post-stabilisation model calibration performance (Henriksen et al, 2008).

Statistical Measure	Abberachrin (whole)		Duvoge (whole)		Lake (immediate)	
	Daily	Monthly	Daily	Monthly	Daily	Monthly
Correlation Coefficient (R)	0.964	0.997	0.997	0.999	0.999	0.999
Nash Sutcliffe (R2)	0.676	0.691	0.885	0.841	0.935	0.919
Statistical Measure	Henriksen Performance Indicator					
	Excellent ★ ★ ★ ★ ★	Very Good ★ ★ ★ ★	Fair ★ ★ ★	Poor ★ ★	Very Poor ★	
Correlation Coefficient (R)	>0.85	0.65-0.85	0.50-0.65	0.20-0.50	<0.20	
Nash Sutcliffe (R2)	>0.85	0.65-0.85	0.50-0.65	0.20-0.50	<0.20	

Figure 6.16 shows the observed and simulated daily and monthly lake level for the *immediate* catchment model over the calibration period. These results demonstrate that the model is generally successful at reproducing the observed daily lake water level despite the flashy nature of the catchment response to precipitation and sluice operation. After the warm up period of the first three months, the model achieves good sequencing of peak water level when the sluice is closed, although the magnitudes of the largest peaks are largely underestimated. The average of the 10 largest peaks in daily water level were simulated as 2% lower than the observed (3.31 mOD (observed); 3.26 mOD (simulated)). In contrast, the troughs in daily water level, when the sluice is open, are overestimated by 9%, especially in March and April 2013 when the model overestimates the daily water level, most notably between 22nd February and 1st April 2013. The most significant overestimation is the magnitude of the peak on 17th April 2013 (2.78 mOD (observed); 3.26 mOD (daily simulated)), suggesting the sluice may not have been fully closed despite the NPWS (National Parks and Wildlife Service) records claiming it was closed by them on 2nd April 2013. The model also overestimates monthly lake level during the latter months in 2013 (March: 2.57 mOD (observed), 2.62 mOD (simulated); April: 2.68 mOD (observed), 2.83 mOD (simulated)).

The statistical measures of model performance confirm the ability of the *immediate* catchment model to simulate daily and monthly lake water level during the calibration period (Figure 6.16 and Table 6.10). According to Henriksen et al (2008) classification, statistical analyses reveal the model is 'Excellent' (R = 0.999; R2 = 0.935) at simulating daily water level, and 'Excellent' (R = 0.999; R2 = 0.919) at simulating monthly water level (Figure 6.16 and Table 6.10).

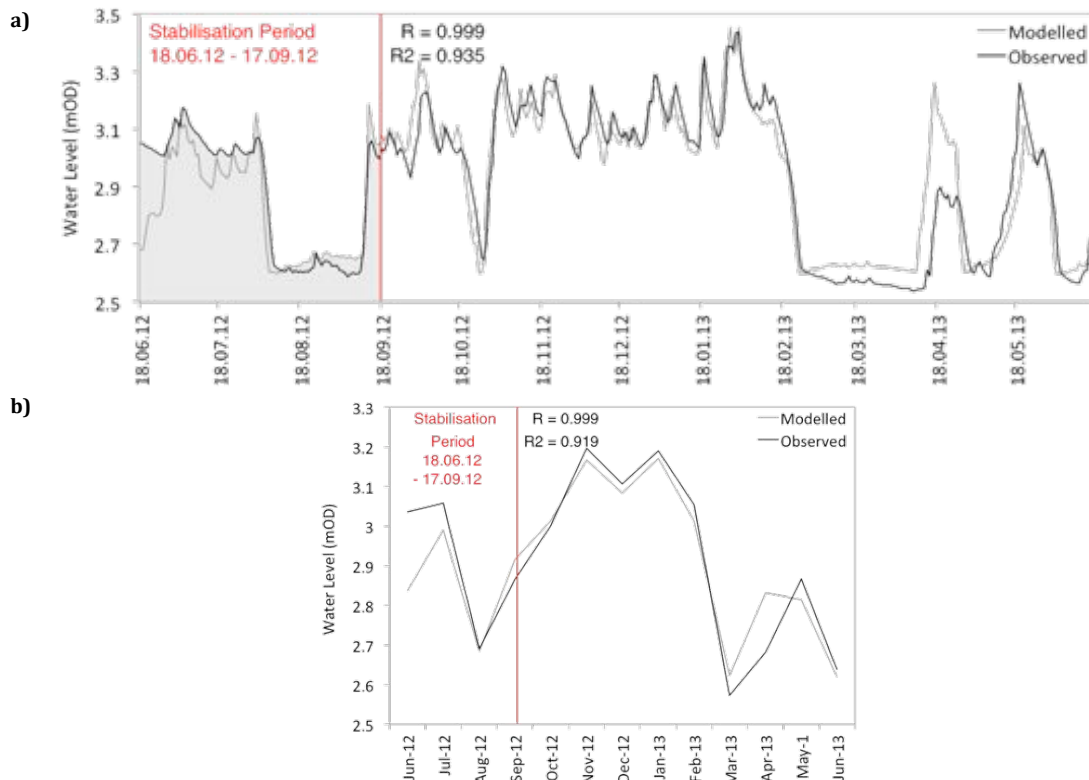


Figure 6.16 Comparison of a) daily and b) monthly observed and simulated lake water level in the *immediate* model for the period of 18th June 2012 to 17th June 2013. Note the model stabilisation period from 18th June to 17th September 2012.

Figure 6.17 shows the observed and simulated Abberachrin and Duvoge daily river discharge for the *whole* catchment model over the validation period. Reasonable agreement between observed and simulated daily discharges is demonstrated. In contrast to calibration, validation results reveal poor sequencing of peak discharge as the magnitudes of most of the largest peaks are overestimated. The average of the 10 largest peaks in daily discharge in each river were simulated as 21% higher than the observed in the Abberachrin ($0.54\text{m}^3\text{s}^{-1}$ (observed); $0.91\text{m}^3\text{s}^{-1}$ (simulated)) and 15% higher than the observed in the Duvoge ($0.60\text{m}^3\text{s}^{-1}$ (observed); $0.87\text{m}^3\text{s}^{-1}$ (simulated)). Although the 10 largest peaks all occur during January, the prolonged period of high discharge in February results in a larger monthly mean. Both of these months are overestimated by the model, specifically by 40% in January and by 31% in February in the Abberachrin, and by 14% in January and by 25% in February in the Duvoge. These modelled discharge overestimations are most likely due to inaccuracies originating from corrections made to the observed meteorological data, specifically when large gaps in the precipitation, temperature and evapotranspiration records (generated due to equipment communication failure) were filled using adjusted regional data (see section 3.3.2 for full details). The troughs in daily discharge are estimated well in the Abberachrin. In the Duvoge, however, troughs during the September to December period are underestimated by 6%, and in March, April and May 2014, the model overestimates the daily discharge by 5%. The model also overestimates the monthly discharge, during these latter months in the Abberachrin.

The statistical measures of model performance indicate the moderate ability of the *whole* catchment model to simulate daily and monthly discharge in the Abberachrin and Duvoge rivers during the validation period (Figure 6.17 and Table 6.11). According to Henriksen et al (2008) classification, statistical analyses reveal the model is only 'Very Good' ($R = 0.954$; $R^2 = 0.709$) at simulating daily Abberachrin river discharge, and 'Very Good' ($R = 0.928$; $R^2 = 0.808$) at simulating daily Duvoge river discharge (Figure 6.17 and Table 6.11). Similarly, the model is only 'Very Good' at simulating monthly river discharge in the Abberachrin ($R = 0.937$; $R^2 = 0.819$) and Duvoge ($R = 0.854$; $R^2 = 0.739$). As discussed above, these statistical results most likely reflect the inaccuracies originating from corrections made to the observed meteorological data.

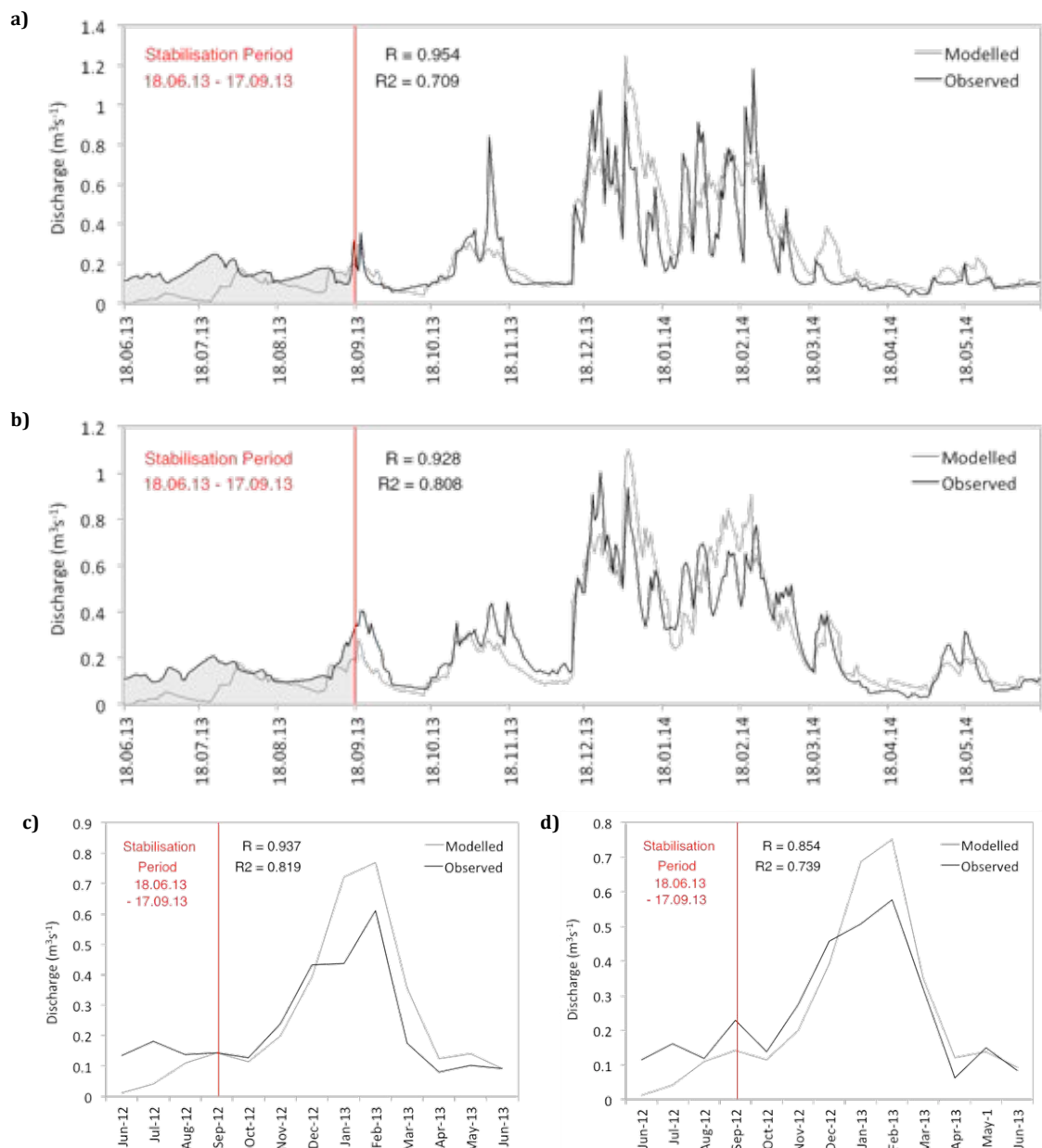


Figure 6.17 Comparison of daily observed and simulated daily discharge in the a) Abberachrin and b) Duvoge rivers in the *whole* model for the period of 18th June 2013 to 17th June 2014. Note the model stabilisation period from 18th June to 17th September 2013.

Table 6.11 Henriksen performance indicators for post-stabilisation model validation performance (Henriksen et al, 2008).

Statistical Measure	Abberachrin (whole)		Duvoge (whole)		Lake (immediate)	
	Daily	Monthly	Daily	Monthly	Daily	Monthly
Correlation Coefficient (R)	0.954	0.937	0.928	0.854	0.798	0.897
Nash Sutcliffe (R2)	0.709	0.819	0.808	0.739	0.689	0.674

Statistical Measure	Henriksen Performance Indicator				
	Excellent ★★★★★	Very Good ★★★★	Fair ★★★	Poor ★★	Very Poor ★
Correlation Coefficient (R)	>0.85	0.65-0.85	0.50-0.65	0.20-0.50	<0.20
Nash Sutcliffe (R2)	>0.85	0.65-0.85	0.50-0.65	0.20-0.50	<0.20

Figure 6.18 shows the observed and simulated lake level for the *immediate* catchment model over the hydrological monitoring survey period. Similar to the modelled river discharge, these results demonstrate a reasonable agreement between the observed and simulated daily lake water level data. In contrast to calibration, the model achieves poor sequencing of peak water level when the sluice is closed as the magnitudes of some of the largest peaks are overestimated. The average of the 10 largest peaks in daily water level were simulated as 7% higher than the observed (3.41mOD (observed); 3.60mOD (simulated)). The troughs in daily water level when the sluice is open are estimated well, except following 18th April 2014, when water levels are overestimated by 21%. The model also overestimates monthly water level, especially for October 2013 (2%), January (37%) and February (20%). These discrepancies are most likely due to the aforementioned inaccuracies originating from corrections made to the observed meteorological data.

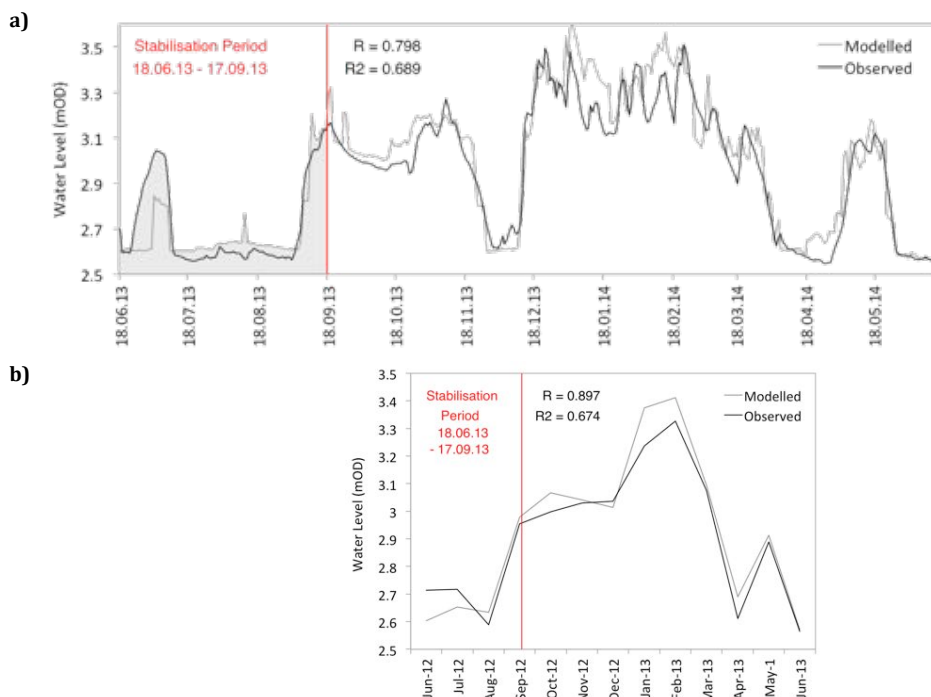


Figure 6.18 Comparison of a) daily and b) monthly observed and simulated lake water level in the *immediate* model for the period of 18th June 2013 to 17th June 2014. Note the model stabilisation period from 18th June to 17th September 2013.

The statistical measures of post-stabilisation model performance indicate the moderate ability of the *immediate* catchment model to simulate daily and monthly lake water level during the validation period (Figure 6.18 and Table 6.11). According to Henriksen et al (2008) classification, statistical analyses reveal the model is only 'Very Good' ($R = 0.798$; $R^2 = 0.689$) at simulating daily water level, and 'Very Good' at simulating monthly water level ($R = 0.897$; $R^2 = 0.684$) (Figure 6.18 and Table 6.11). Again, these statistical results most likely reflect the inaccuracies originating from corrections made to the observed datasets.

6.4 Climate change impacts

Climate change during the 21st century is likely to have important consequences for the conservation of freshwater habitats (Kundzewicz et al, 2007; Bates et al, 2008; Thompson 2012). Projections for future climate in the UK and Ireland, such as those produced by the UK Climate Impacts Programme (UKCIP) for the region of northwest Ireland (derived from Water Framework Directive river basin regions), are characterised by hotter drier summers, warmer wetter winters and more frequent and intense precipitation events (Hulme et al, 2002; Murphy et al, 2009). Similarly, a report on future Irish climate conditions by Ireland's Environmental Protection Agency reveals that, by 2080, a general warming of between 1.25°C and 1.5°C is likely (McGrath et al, 2005). For precipitation, the most significant changes are predicted to occur in June, when precipitation is likely to decline, and in December, when precipitation is expected to increase. Finally, the report presents evidence of increases in the frequency of extreme precipitation events (i.e. $>20\text{mm day}^{-1}$) and also of intense storm events across northwest Ireland. It should be noted, however, that the overall predicted trends of warming and increases in precipitation and evapotranspiration outlined in these reports are likely to be complicated by inconsistent seasonal and annual effects and year-to-year variation.

As discussed in section 1.2.3, freshwater systems are sensitive environments that are vulnerable to climate change (Thompson et al, 2004). The growing appreciation of freshwater systems such as lakes and wetlands as important ecosystems for both wildlife and humans over the last decades of the 20th century has led to recent acknowledgment that their loss and degradation is a major cause for concern (Williams, 1990; Thompson & Hollis, 1997; Mitsch & Gosselink, 2000; Finlayson & Moser, 1992; Mitsch et al, 1994; CEC, 1995). In response to such concerns, a host of initiatives have been developed at the local, national and international level, that aim to improve the management of existing freshwater systems and restore or recreate those that have been lost or degraded.

This chapter investigates the impacts of projected climate change on the Sheskinmore Lough lake and wetland system using the previously developed MIKE SHE / MIKE 11 models. Climate change scenarios were developed using the UKCIP09 probabilistic projections and used to perturb the meteorological inputs to these models. Results are compared to baseline conditions provided by the model for the calibration and validation periods.

6.4.1 Development of climate change scenarios

Following the approach described by Thompson (2012) the impacts of climate change on Sheskinmore Lough were assessed in three stages. Model calibration and validation described above comprised the first stage. Subsequently, a series of climate change scenarios were defined and the original input meteorological data perturbed accordingly. The hydrological models were then run using the perturbed meteorological data and simulated lake level and above ground water depth compared to the baseline conditions provided by the original model. Other model parameters such as those representing land cover and geology remained unchanged, an approach that is widely used in hydrological modelling assessments of climate change (Fowler & Kilsby, 2007; Johnson et al, 2009; Kingston et al, 2010; Singh et al, 2010; Thompson, 2012).

Perturbations to model atmospheric variables (in this case precipitation and temperature) were provided by the 2009 UK Climate Projections (UKCP09) following the approach used in similar modelling studies including Bell et al (2012), Thompson (2012), and Afzal et al (2015). UKCP09 provides probabilistic projections for a number of atmospheric variables under three emissions scenarios, referred to as 'Low', 'Medium' and 'High', which correspond to the B1, A1B and A1FI scenarios, respectively, in the IPCC Special Report on Emission Scenarios (SRES) (IPCC, 2000; Jenkins et al, 2009). Projections for atmospheric variables are provided in the form of a probability distribution function designed to represent future climate uncertainties (Thompson, 2012). The methodology used to generate these projections is based on a large perturbed physics ensemble using the Met Office Hadley Centre's HadCM3 global climate model and results of another set of 12 global climate models (IPCC, 2000). Projections are downscaled to a resolution of 25km² using the HadRM3 regional climate model and are provided for a series of seven overlapping 30-year time slices: 2010–2039, 2020–2049, 2030–2059, 2040–2069, 2050–2079, 2060–2089 and 2070–2099 (Murphy et al, 2009). Changes in atmospheric variables are available for monthly, seasonal and annual average periods and are expressed relative to a 30-year baseline period (1961–1990).

In this thesis, projections for the three emissions scenarios for the 2040–2069 and 2070–2099 time periods (referred to as the 2050s and 2080s, respectively) were selected to represent conditions towards the middle and end of the current century. As described by Thompson (2012) and Christerson et al (2012), these time periods were chosen due to their relevance for conservation management planning in the long-term. Monthly changes in precipitation (%) and mean temperature (°C) were abstracted for probabilities between the 10 and 90% levels in 20% increments from the HadRM3 model area for the region of northwest Ireland delineated in Figure 6.19. As suggested by Murphy et al (2009), probabilities outside of this range were not used as they are deemed extreme and therefore more likely to produce unrealistic projections. The range of probabilities employed includes the central estimate of change (i.e. the change that is as likely as not to be exceeded: 50% probability level), bounded by the changes that are very likely to be exceeded (10% probability level) and those that are very unlikely to be exceeded (90% probability level). A total of 30 scenarios were developed comprising five different

probabilities for each of the three emissions scenarios across the two time slices. This thesis will refer to these scenarios using abbreviations: e.g. 2050L₁₀ (the 2050 time slice, Low emissions scenario (A1B), 10% probability level).



Figure 6.19 Coverage of UKCIP 2009 probabilistic projections for future climate in the river basin region of northwest Ireland.

The corrected daily precipitation data recorded by the Sheskinmore Lough weather station were multiplied by the UKCP09 monthly percentage change factors for each respective month to provide new time series for each scenario. Similarly, projected change factors for mean temperature were added to the observed weather station-derived temperature data and then Penman-Monteith evapotranspiration was recalculated (Zotarelli et al, 2010). Given the comparatively moderate validation results, most likely due to inaccuracies introduced to the observed meteorological data during the second-year of the monitoring (see explanation of weather station problems in section 3.3.2), only the first year (18th June 2012 to 17th June 2013) of the hydrological monitoring period was assessed going forwards. Simulated climate change results were compared with those for the same period. As the simulation period falls outside the 30-year UKCP09 baseline period (1961-1990), results are likely to be representative of conditions at the latter part of each time slice (Thompson et al, 2009; Thompson, 2012).

6.4.2 Modelled hydrological response to climate change

Changes in the distribution of monthly mean absolute precipitation and evapotranspiration for each of the emissions scenarios for the 2050s and 2080s periods are summarised in Figure 6.20. The perturbed data reveal increases in the magnitude of changes with progressively higher emissions scenarios across the two time periods. Changes in precipitation are predominantly larger than those for evapotranspiration. At the 70% probability level for all three emissions scenarios within both time slices, precipitation increases in all months except for June. The largest increases in both

absolute and percentage terms occur in December, the second wettest month under baseline conditions (321.3mm), and vary between 36.9mm (10.3%) under the 2050L50 scenario and 52.5mm (14.1%) under the 2050H50 scenario, and 61.6mm (16.1%) and 101.8mm (24.1%) for the 2080L50 and 2080H50 scenarios, respectively. Percentage changes in total precipitation from January to June are less than half of those for December. The reductions in precipitation for the months June to October and from April to June are large, varying between reductions of 17.7% (2050L₁₀) and 23.9% (2050H₁₀) for the 2050s, and 25.7% (2080L₁₀) and 29.8% (2080M₁₀). At progressively higher probability levels the number of months in which precipitation is projected to increase, as well as the magnitude of increases when they occur, both increase. Within all the emissions scenarios and both time slices the 10% probability level only produces increases in precipitation for one month (December) with the exception of 2050L₁₀, which results in a reduction in mean monthly precipitation for every month. In contrast, at the 90% probability level, every month experiences increases in precipitation which are as much as an additional 140.2mm (30.4%) for 2080L₉₀ and 208.8mm (39.4%) for 2080H₉₀.

As expected, evapotranspiration increases in every month across all scenarios with the largest absolute increases occurring between September and April. In percentage terms, however, the differences between changes in evapotranspiration in these and other months are generally smaller. The magnitude of gains in evapotranspiration increases with probability level and across both scenario time periods. For example, under baseline conditions the monthly evapotranspiration in June is 35.4mm. Under the 2050s Medium emissions scenario this increases slightly by 1.9mm (5.09%) for the 10% probability level: 2050M₁₀. The corresponding increases for the 2050M₃₀, 2050M₅₀, 2050M₇₀ and 2050M₉₀ scenarios are 3.4mm (8.76%), 4.5mm (11.3%), 5.5mm (13.4%) and 6.9mm (16.3%), respectively. For all probability levels, changes in evapotranspiration are larger with the progressively higher emissions scenario. For example, changes for December under 2050L₅₀ are 9mm (19.5%) compared to 10mm (21.2%) for 2050M₅₀ and 10.6mm (22.4%) for 2050H₅₀. The 2080s time period is associated with larger changes than the 2050s time period. For example, December evapotranspiration under 2080M₅₀ increases by 11.9mm (24.3%).

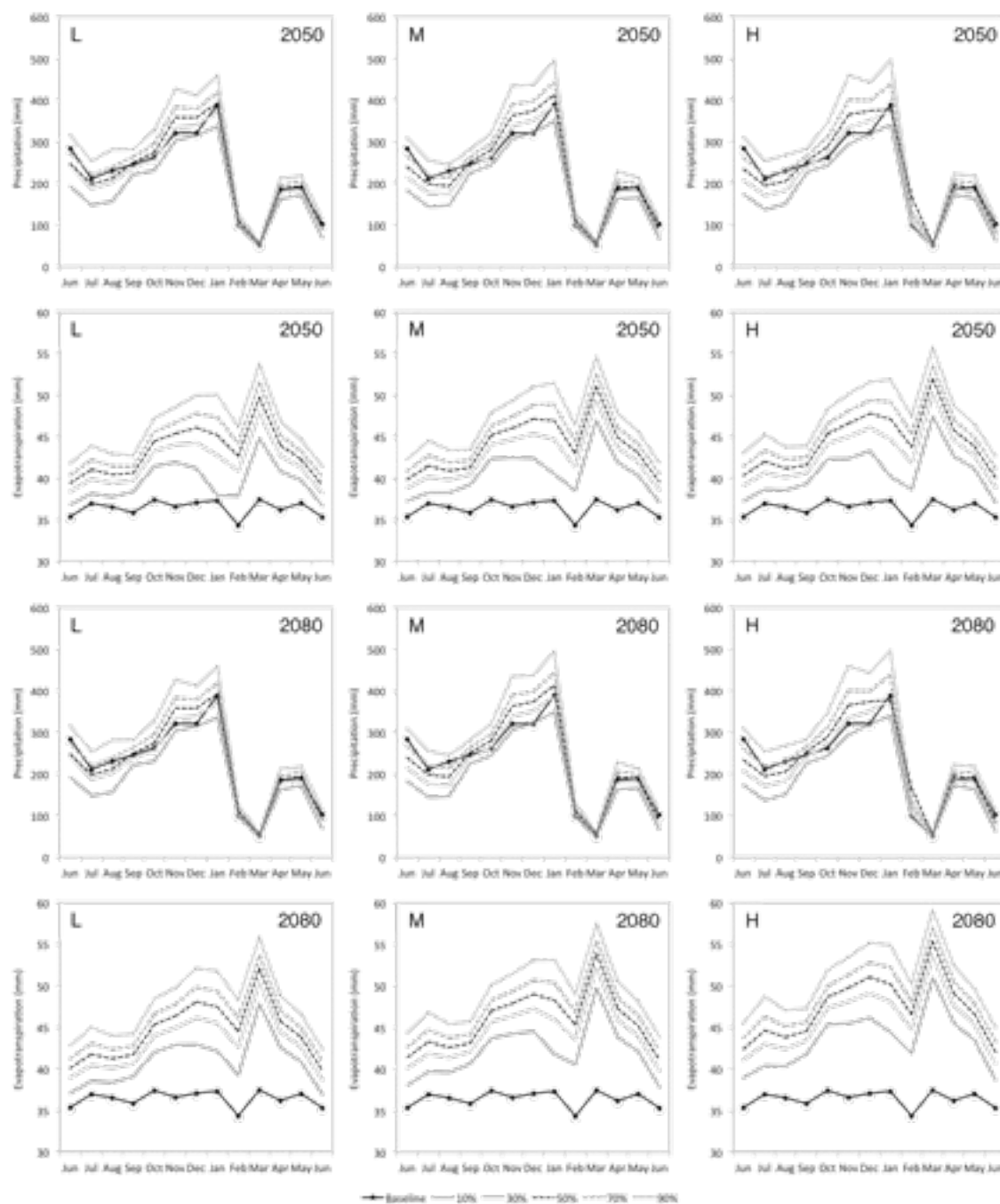


Figure 6.20 Monthly mean absolute catchment precipitation and evapotranspiration for the baseline scenario at probabilities between 10% and 90% for each emissions scenario (L = Low, M = Medium, H = High) for the 2050s and 2080s.

The 2050 to 2080 trends identified in Figure 6.20 are summarised in Figure 2.21, which shows mean absolute annual, summer (defined by the UKCP09 as June–August) and winter (December–February) precipitation and evapotranspiration for all the scenarios. Mean annual precipitation is very likely to increase (70% and 90% probability levels) relative to the baseline (2689.8mm) at all three emissions levels in 2050 and 2080. At the 10% and 30% probability levels, however, projected precipitation below the baseline in both 2050 and 2080 is more likely than not to be exceeded. Central estimates of change for the 2050s are declines of between 89.3mm (3.32%) and 390mm (14.5%) and

increases of between 52.4mm (1.95%) and 690mm (25.6%). For the 2080s, however, central estimates of change (50% probability level) are declines of between 52.4mm (1.95%) at 2050L₅₀ and 119.2mm (4.43%) at 2050H₅₀. There is, however, an overwhelming trend towards enhanced winter precipitation. Declines during this season are limited to the 10% probability level for all scenarios and all probability levels for the High emissions scenario in 2050. Conversely, summer precipitation is primarily projected to decline in 2050 and 2080 across the 10% to 70% probability levels. Exceptions are projected summer precipitation increases under the 90% probability level and under the High emissions scenario in 2050. In percentage terms, changes in spring (March-May) and autumn (September–November) precipitation (not shown) are similar to those for annual totals.

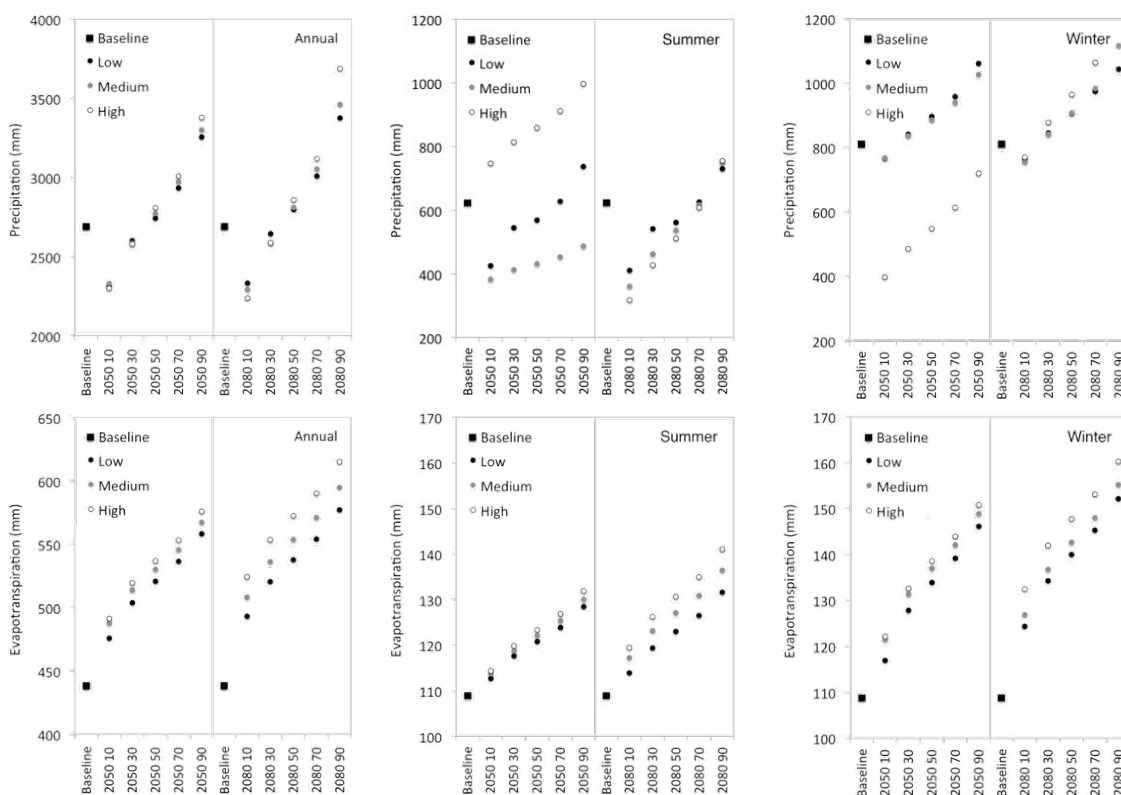


Figure 6.21 Projected mean absolute annual, summer and winter catchment precipitation and evapotranspiration for the baseline scenario at probabilities between 10% and 90% for each emissions scenario (Low, Medium, High) for the 2050s and 2080s. Note the different y-axis scales.

In contrast to the variations in projected precipitation, evapotranspiration increases at all of the probability levels and emission scenarios for both the 2050s and 2080s time slices. Annual evapotranspiration, which under baseline conditions is 438.2mm, is by the 2050s very likely to exceed (10% probability level) between 475.9mm (8.59% increase, 2050L₁₀) and 491.3mm (12.1% increase, 2050H₁₀) and very unlikely to exceed (90% probability level) between 558.1mm (27.3% increase, 2050L₉₀) and 575.7mm (31.4% increase, 2050H₉₀). Central estimates of change (50% probability level) for the 2050s are increases of between 82.4mm (18.8%) at 2050L₅₀ and 98.5mm (22.5% increase, 2050H₅₀). By the 2080s these changes increase so that for the central estimates of change annual evapotranspiration is between 99.6mm (22.7% increase, 2080L₅₀) and 134.1mm (30.6%

increase, 2080H₅₀). Summer and winter evapotranspiration reflect the annual trend, with both increasing above the baseline in all projection scenarios. The likely increases in winter evapotranspiration, however, display a much larger range than those projected for the summer in both the 2050 and 2080 scenarios. In similar fashion to precipitation, the projected increases in spring (March-May) and autumn (September-November) evapotranspiration (not shown) are similar to those for annual totals.

Figure 6.22 shows the mean daily water level in Sheskinmore Lough abstracted from simulation results for the baseline and each emissions scenario for the 2050s and 2080s. Climate change results are shown for probabilities between 10% and 90% in 20% increments. Throughout the majority of the modelling period mean daily water levels increase, by 9% up to a maximum of 3.71mm under the 2050M₉₀ emissions scenario and by 22% up to a maximum of 3.98mm under the 2080L₉₀ emissions scenario. The increases are primarily associated with periods of sluice closure (i.e. when water levels are around 2.6mOD). When the sluice is open (i.e. when water levels are around 3.2mOD), however, variability around the baseline is minimal at all probability levels and under all emissions scenarios, with comparatively small increases (maximum 22%, 2080L₁₀) and decreases (maximum 3%, 2050M₅₀) in mean daily water level. During sluice closure, the amplitude of projected water level fluctuations increases by an average of 28% under the low emissions scenario and by 35% under the high emissions scenario in 2050, and by 40% under the low emissions scenario and 30% under the high emissions scenario in 2080.

The mean monthly water level in Sheskinmore Lough abstracted from simulation results for the baseline and each emissions scenario for the 2050s and 2080s is shown in Figure 6.23. On the whole, simulated changes in mean water level closely follow the monthly projected changes in precipitation discussed above. From Low to High emissions scenarios there is a general positive trend towards higher projected mean water levels throughout the year in both 2050 and 2080 (Figure 6.24). There is, however, more uncertainty associated with projected monthly, annual and seasonal water levels in 2080.

For the 2050s time slice, mean annual water levels increase up to a maximum of 78mm (2.7%) under the 2050H₉₀ emissions scenario and up to 128mm (4.4%) under the 2080L₉₀ emissions scenario. This trend is interrupted, however, by more suppressed increases in mean monthly water levels under the Medium emissions scenario at the 50%, 70% and 90% probability levels in both 2050 and 2080, most likely due to reduced monthly precipitation projected by the Medium emissions scenario at these higher probability levels. An exception to this positive trend occurs during June, July and August when water levels are likely to decrease by up to 10cm at the 10% and 30% probability levels under all emissions scenarios for the 2050s and 2080s time slices. Indeed, annual water levels in 2080 are more than likely to decrease at the 10% and 30% probability levels under the Low and Medium emissions scenarios. In 2050, these probability levels are likely to produce small changes relative to the baseline water level under all three emissions scenarios. The ranges in mean autumn and winter water levels projected for both 2050 and 2080, are likely to be of greater magnitude than those projected for the spring and summer months at all probability levels and under each emissions scenario.

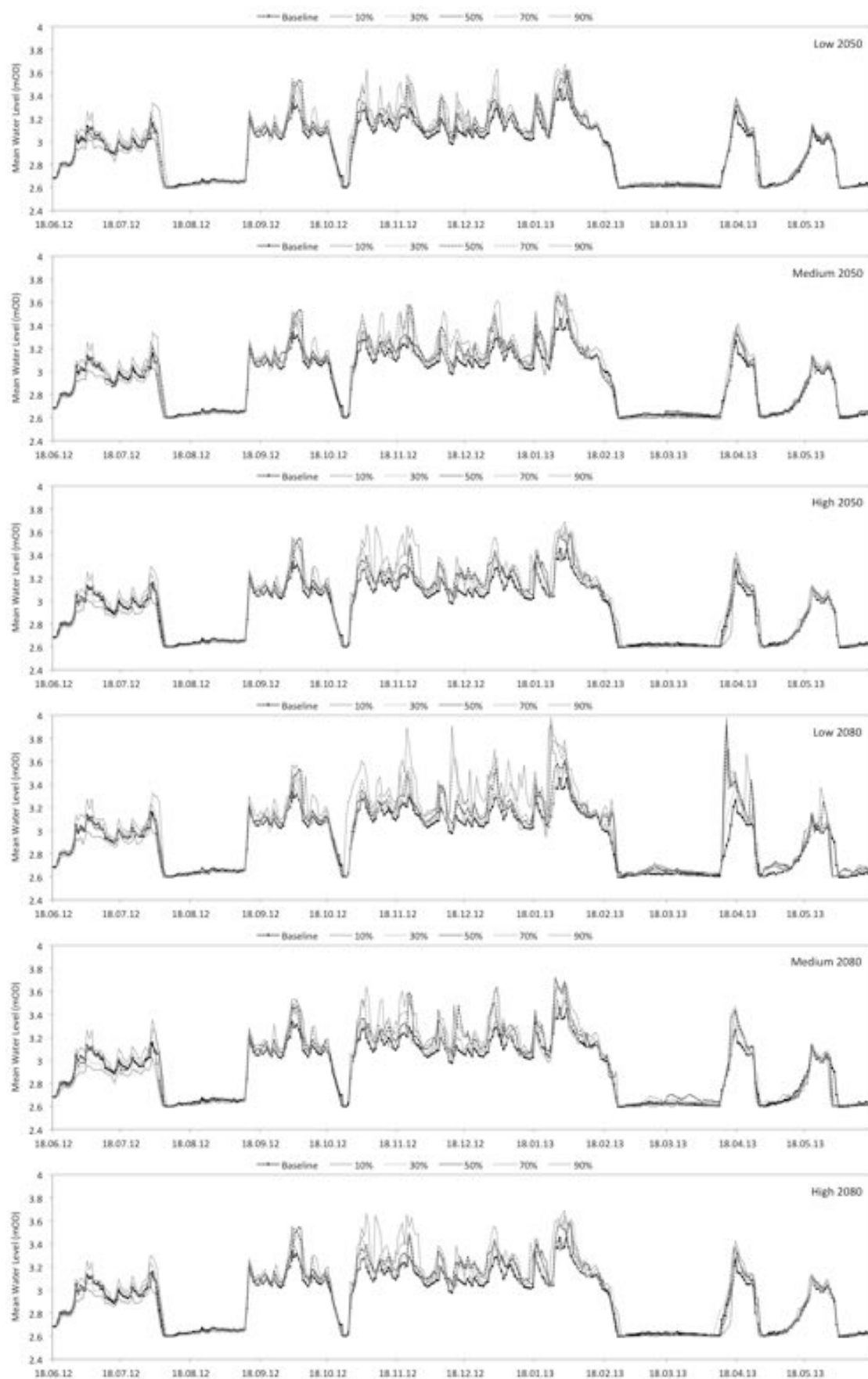


Figure 6.22 Projected mean daily water level in Sheskinmore Lough for the baseline scenario at probabilities between 10% and 90% for each emissions scenario (L = Low, M = Medium, H = High) for the 2050s and 2080s.

Mean monthly water level is projected to increase to the greatest degree in November, specifically by 255mm (8.1% increase) at the 2050H₉₀ scenario and by 288mm (9.1% increase) at the 2080H₉₀ scenario, and also in December by 192mm (6.2% increase) at the 2050M₉₀ scenario and by 279mm (9.1% increase) at 2080H₉₀. In 2080, mean monthly water level is as likely as not to increase significantly in April (up to 6% increase at 2080L₅₀). Mean monthly water level is, however, projected to decrease during the majority of the spring and summer months. The largest decreases are projected in June by 279mm (5.1% decrease, 2050H₁₀), and in July by 293mm (6.4% decrease, 2050H₁₀) and 288mm (11.3% decrease, 2080H₁₀), however these are very likely to be exceeded. In March, smaller decreases are projected by 260mm (2.6% decrease, 2050M₉₀) and 278mm (3.4% decrease, 2080M₉₀), however they are very unlikely to be exceeded.

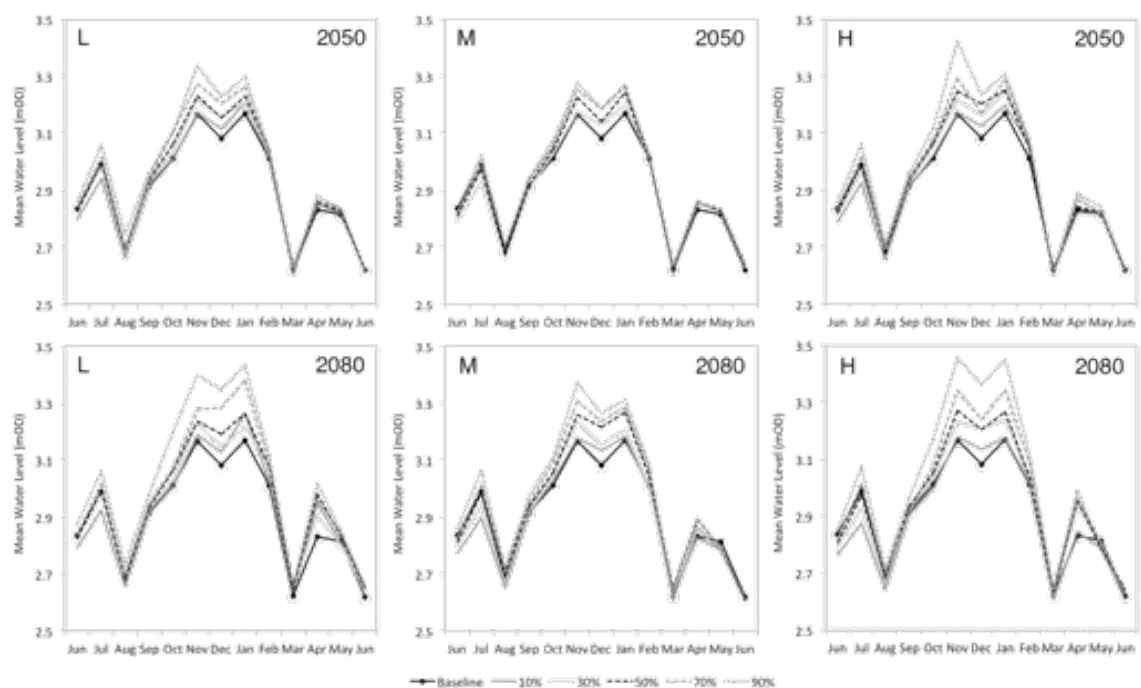


Figure 6.23 Projected mean monthly water level in Sheskinmore Lough for the baseline scenario at probabilities between 10% and 90% for each emissions scenario (L = Low, M = Medium, H = High) for the 2050s and 2080s.

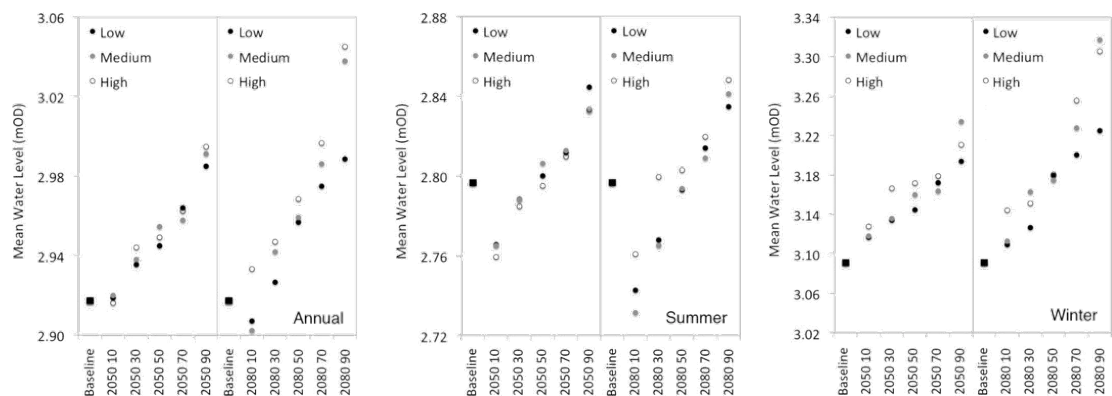


Figure 6.24 Projected mean annual, summer and winter water level in Sheskinmore Lough for the baseline scenario at probabilities between 10% and 90% for each emissions scenario (Low, Medium, High) for the 2050s and 2080s. Note the different y-axis scales.

The potential impacts of future climate change on overland water depth across the Sheskinmore Lough lake and wetland system are displayed in Figure 6.25. The time steps chosen were based on the maximum (Max), minimum (Min) and mean (Mean) daily lake water level modelled under each emissions scenario (Low, Medium and High) for the 2050s and 2080s. The maximum (Max), minimum (Min) and mean (Mean) daily lake water levels modelled under baseline climate conditions are also provided as baseline overland water depth. It should be noted that the modelled baseline reveals the model consistently overestimates the depth of overland water at the western end of the wetland, by as much as 70cm. This is most likely due to a combination of inaccuracies in the post-processed topographical data and errors in the pre- and post-processed saturated, and unsaturated, zone data resulting from a lack of detailed geological information. The results do, however, show some distinctive patterns in the behaviour of overland water depth in response to future climate change.

In 2050, under all emissions scenarios, overland water depth in the lake area is projected to reach 1.60m when water levels are at their maximum, especially under the Medium emissions scenario. It should be noted that these maximums, simulated at the 90% probability level, are very unlikely to be exceeded. At these maximum levels, overland water depth in the reedbed is projected to reach an average of 80cm, indicating flooding across the entire habitat area. Taking modelled overestimations into account, flooding is also likely to occur in the wetland, albeit only at depths of up to 20cm to 30cm. In contrast, when water levels are at their minimum, under the Low emissions scenario overland water depths in the lake closely resemble those under baseline climate conditions. Under the Medium and High emissions scenarios, water depths in the lake are projected to decline, especially under the High emissions scenario, where minimum water depths are predicted to reach as low as 10cm to 20cm in the deepest areas. Mean overland water depths in the lake are projected to reach their highest levels (1m) under Medium emissions, in comparison to depths of 80cm and 50cm under High and Low emissions, respectively. The coarse resolution of the modelled data, however, means any minor undulations, which are important topographical features in such a shallow lake, are very likely to be masked. The deeper areas of the lake and reedbed are therefore likely to experience minimum water depths of 30cm depth under the High emissions scenario.

In 2080, overland water depth is projected to increase with the increasing emissions scenarios, with the greatest increases predicted under the High emissions scenario. Mean water depths, however, are also projected to increase, primarily under Medium and High emissions. Under Low emissions, there is very little difference between the mean and maximum projected overland water depth, with maximums of 80cm depth within the lake. In contrast to 2050, minimum water depths under the Low and Medium emissions scenarios in 2080 are predicted to be 10cm to 20cm below baseline conditions. It should be noted, however, that these projections were made at the 10% probability level and are therefore very likely to be exceeded. Under the High emissions scenario, overland water depths in the lake and reedbed in 2080 are expected to reach maximums of 1.50m and 1.60m depth, respectively; however these depths are very unlikely to be exceeded. Within the wetland, flooding (30cm to 40cm) is also likely under Medium and High emissions.

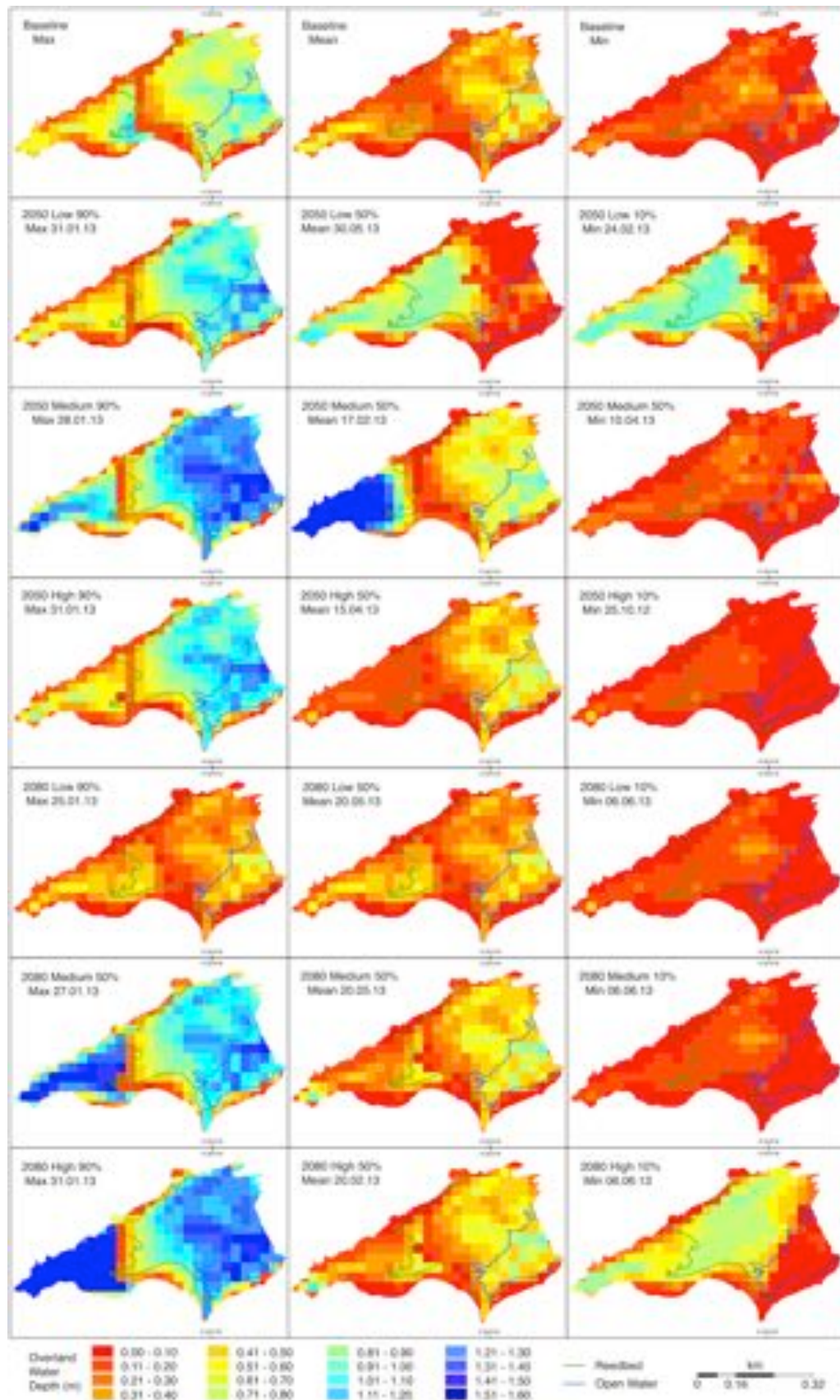


Figure 6.25 Impacts of future climate change on overland water depth (m) across the Sheskinmore Lough lake and wetland system for a range of scenarios based on the maximum (Max), minimum (Min) and mean (Mean) modelled lake water level under each emissions scenario (Low, Medium and High) for the 2050s and 2080s. Note the lake open water and reedbed habitats are outlined in blue and green, respectively, while the remainder of the modelled area encompasses the wetland habitat.

6.5 Hydrological management impacts

As discussed in section 1.2.3, freshwater habitat loss and degradation over the last few decades has led to the development of a whole host of hydrological management initiatives, from site to international levels. Hydrological management initiatives aim to improve the management of existing freshwater systems, restore lost habitats and rejuvenate those that have become degraded. Understanding the impacts of these initiatives and improving the ability to predict them in the future are therefore required in order to develop management schemes that will achieve their goals, avoid undesirable outcomes and effectively target the often limited resources available to wetland management and conservation practitioners.

6.5.1 Selection of management scenarios

Given the importance of hydrological management at Sheskinmore Lough and the observed dramatic influence of the sluice on contemporary lake water levels (see section 4.3.2.1 for more details) it was considered appropriate to simulate alternative sluice management scenarios by varying the level of the sluice. As the sluice can either only be set to *fully open* or *fully closed* (i.e. it cannot be adjusted to any other level) (section 4.4.2) these two sluice scenarios were chosen as the most realistic and effective way to assess hydrological management via modelling. The management scenarios are based on three possible decisions that could be made by water managers at the site: the sluice varied *as is* (i.e. opened and closed) in response to changing water levels; the sluice left *fully open* at all times; and the sluice left *fully closed* at all times.

6.5.2 Modelled hydrological response to management decisions

Mean daily water level in Sheskinmore Lough abstracted from simulation results for 2050s and 2080s under the three management scenarios is shown in Figure 6.26. The modelled scenarios clearly show the influence of the sluice on lake water levels under baseline climate conditions. As observed in section 4.4.2.1, the *as is* scenario exhibits large shifts in water level from an average level of 3.1mOD when the sluice is closed to an average level of 2.6mOD when the sluice is open. Under the *fully open* scenario, water levels are maintained at the lower (2.6mOD) average elevation throughout the modelling period and only vary by a maximum of 10cm. In contrast, when fully closed, water levels are maintained at the higher (3.1mOD) average elevation; however they do fluctuate around this average elevation between 2.9mOD and 3.7mOD. Although the *fully closed* scenario largely mirrors the *as is* scenario water levels when the sluice is closed, the *fully closed* scenario exhibits larger increases in water level fluctuations during discrete points during the modelling period on 28th January 2013, 17th-18th April 2013 and 19th-20th May 2013.

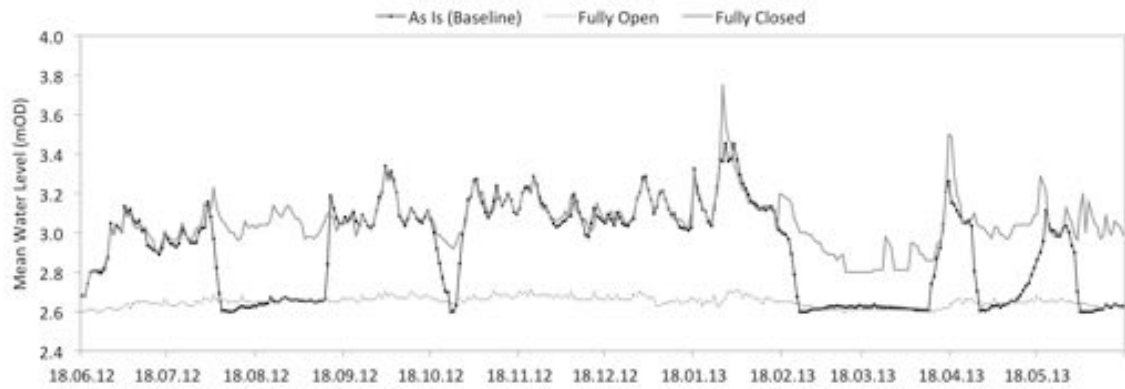


Figure 6.26 Projected mean daily water level in Sheskinmore Lough for the *as is*, *fully open* and *fully closed* scenarios under the baseline (*as is*) climate conditions.

Modelled impacts of the hydrological management scenarios on overland water depth across the Sheskinmore Lough lake and wetland system are displayed in Figure 6.27. As discussed in the previous section, due to data inaccuracies, the model overestimates overland water depth at the western end of the wetland. In comparison to the *as is* scenario, when *fully closed* overland water depths are noticeably enhanced, with mean water depths in the lake reaching 80cm to 90cm in the deepest areas in comparison to 70cm to 80cm under the *as is* scenario. When *fully open*, however, mean water depths in the lake fall to below 60cm, exposing the majority (80%) of the lake area to depths of under 20cm, with depths of 0cm at the southern lake fringe. Maximum water depths are projected to be greatest under the *fully closed* scenario (up to 1.60m) and minimum water depths lowest (<20cm across 90% of the lake area) under the *fully open* scenario.

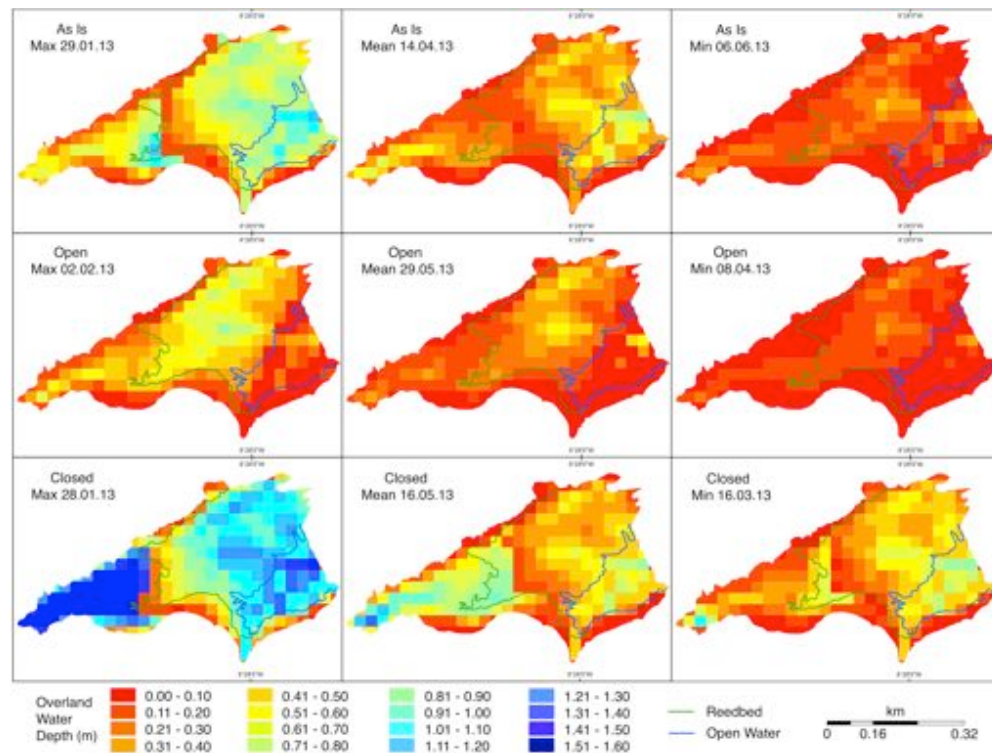


Figure 6.27 Impacts of the *as is* (As Is), *fully open* (Open) and *fully closed* (Closed) management scenarios on maximum (Max), minimum (Min) and mean (Mean) overland water depth across the Sheskinmore Lough lake and wetland system under current climate conditions. Note the lake open water and reedbed habitats are outlined in blue and green, respectively, while the remainder of the modelled area encompasses the wetland habitat.

6.5.3 Modelled hydrological response to management decisions and climate change

Mean monthly water level in Sheskinmore Lough abstracted from simulation results for the *initial* management scenarios under current climate conditions, and the management scenarios under future climate change emissions in 2050s and 2080s are summarised in Figure 6.28. The climate change projections are made relative to the respective *initial* management scenarios, which were modelled under current climate conditions. Results are shown for probabilities at the 10%, 50% and 90% levels only, as previous analyses have revealed the intervening probability results (30% and 70%) lie approximately mid-way between those of the adjacent probabilities. In comparison to the *as is* management scenario, under which mean monthly water levels vary by up to 0.55m in the current climate and by up to 0.82m in 2080 (H_{90}), mean monthly water levels under the *fully closed* scenario vary by up to 0.39m in the current climate and by up to 0.60m in 2080 (H_{90}) (Table 6.12). The *fully open* scenario, however, exhibits an even smaller mean monthly water level range only, varying by up to just 0.07m in the current climate and by up to 0.08m in 2080 (H_{90}). These ranges are, however, based on future climate change projections at the 90% probability level and are therefore very unlikely to be exceeded. Nevertheless, even at lower probability levels, the contrast in water level range between the *as is* and *fully closed* scenarios and the *fully open* scenarios is still apparent.

Under future climate change scenarios, the *fully closed* projections exhibit similar seasonal patterns of response to those under the *as is* scenario, with water levels increasing in the winter months and decreasing slightly at the 10% probability level during the summer. It should be noted, however, that this summer decrease is very likely to be exceeded at the 10% probability level. In contrast to the *as is* scenario, the pattern of winter increases and summer decreases is more pronounced under the *fully closed* scenario. For example, during the winter (November to January), under the *as is* scenario water levels increase by an average of 0.28m, however under the *fully closed* scenario they increase by an average of 0.35m. In contrast, under the *fully open* scenario, mean monthly water levels are generally projected to decline, albeit only very slightly during the summer, autumn and winter at the 10% probability level. Little change is projected at the 50% probability level and only at the 90% probability level are water levels projected to increase, primarily during the winter and spring. Manipulating the sluice arguably has the largest impact on mean monthly lake water level range in comparison to the influence of climate change.

As well as mean monthly water level ranges Table 6.12 also details mean daily water level ranges under initial and future climate for the three management scenarios. Under the *as is* management scenario, in the current climate, mean daily water levels range by 0.85m; however under the *fully closed* scenario this range extends to 0.92m and under the *fully open* scenario is reduced to 0.10m. Under future climate change, these water level ranges are enhanced, however these increases are not consistent with increasing emissions. Overall, the High emissions scenario produces the largest increases in water level ranges, with a maximum range of 1.25m in 2080 under the *as is* management scenario. In contrast, the Medium emissions scenario produces the smallest increases in water level ranges, with a maximum range of 1.12m in 2080 under the *as is* management scenario.

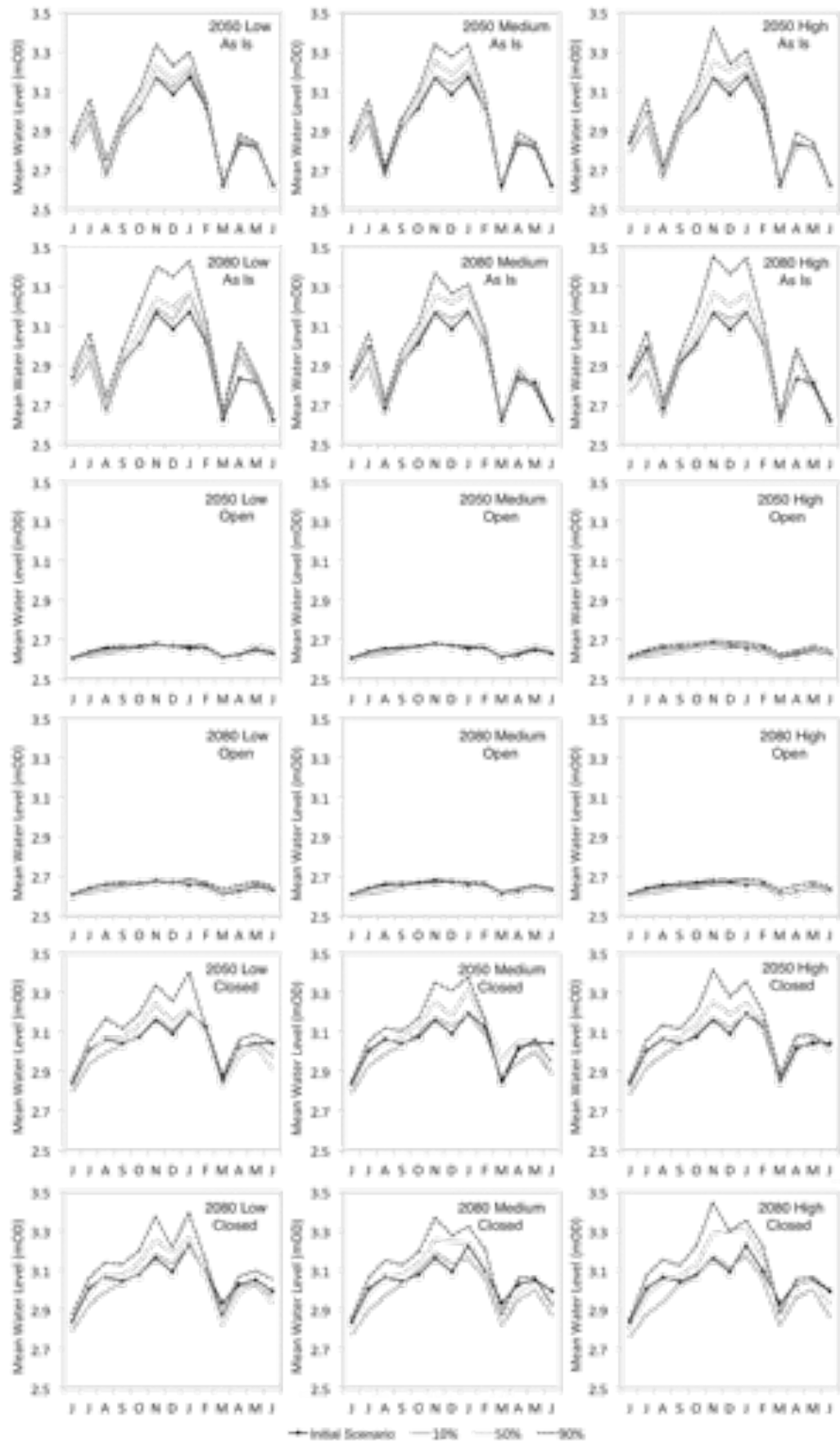


Figure 6.28 Projected mean monthly water level in Sheskinmore Lough under the three *initial* management scenarios (*as is*, *fully open*, *fully closed*) in the current climate, and also under the future climate change emissions scenarios (L = Low, M = Medium, H = High) at probabilities between 10% and 90% for the 2050s and 2080s.

Table 6.12 Range of projected mean daily and monthly lake water levels under each *initial* (current climate) scenario and also under each emissions scenario (L = Low, M = Medium, H = High) across the range of management scenarios (*as is*, *fully open* and *fully closed*), at probabilities between 10% and 90% for the 2050s and 2080s.

Management Scenario	Initial (Current Climate) Scenario Daily (& Monthly) Water Level Range (m)	Probability Level (%)	Daily (& Monthly) Water Level Range (m)					
			2050 Emissions Scenario			2080 Emissions Scenario		
			Low	Medium	High	Low	Medium	High
<i>As Is</i>	0.85 (0.55)	10	1.04 (0.60)	0.98 (0.57)	0.98 (0.59)	1.32 (0.63)	0.93 (0.58)	1.00 (0.57)
		50	1.02 (0.62)	1.06 (0.63)	0.99 (0.64)	1.32 (0.62)	1.12 (0.66)	1.15 (0.64)
		90	1.07 (0.73)	1.11 (0.74)	1.09 (0.82)	1.38 (0.78)	1.09 (0.76)	1.25 (0.82)
<i>Fully Open</i>	0.10 (0.07)	10	0.11 (0.07)	0.11 (0.07)	0.10 (0.07)	0.10 (0.07)	0.13 (0.07)	0.12 (0.06)
		50	0.13 (0.08)	0.14 (0.07)	0.13 (0.07)	0.15 (0.07)	0.13 (0.07)	0.16 (0.08)
		90	0.14 (0.07)	0.17 (0.07)	0.17 (0.07)	0.14 (0.08)	0.14 (0.07)	0.16 (0.08)
<i>Fully Closed</i>	0.92 (0.39)	10	0.94 (0.40)	0.86 (0.40)	1.00 (0.40)	0.98 (0.43)	0.83 (0.41)	0.84 (0.41)
		50	0.94 (0.43)	1.20 (0.50)	0.94 (0.44)	0.93 (0.45)	1.00 (0.45)	1.05 (0.51)
		90	1.09 (0.54)	1.08 (0.54)	1.21 (0.56)	1.13 (0.53)	1.07 (0.51)	1.20 (0.60)

Impacts of the combined effect of future climate change (Medium emissions scenario only) and hydrological management (*as is*, *fully open* and *fully closed*) on overland water depth across the Sheskinmore Lough lake and wetland system are shown in Figure 6.29. Under the *as is* scenario and Medium emissions, in 2050 mean water depths reach 1m in the deepest parts of the lake. In 2080, however, mean water depths fall slightly under the *as is* scenario, to 90cm in the deepest areas. This decline is mirrored in the projected mean water depths under the *fully closed* scenario, which exhibits a mean depth of up to 90cm in 2050 and up to 80cm in 2080 in the deepest areas of the lake. Under the *fully open* scenario, however, mean water depths in the deepest areas of the lake increase slightly, from up to 30cm in 2050 to up to 40cm in 2080.

Mean overland water depths under the *as is* and *fully closed* scenarios are considerably higher (70cm) than those projected under the *fully open* scenario. Maximum water depths are greatest under the *as is* scenario, with depths of up to 1.60m in the deepest areas of the lake. Under these conditions, water depths are predicted to result in flooding of the entire lake and reedbed area in 2050 and 2080, with depths of 1m projected across the majority of the habitats; however at the 90% and 50% probability levels, these depths are very unlikely and equally likely to be exceeded, respectively. Future flooding is also predicted within the wetland, albeit at lower levels (maximum depths of 20cm to 30cm) than projected. Similarly, under the *fully closed* scenario, maximum water depths are predicted to result in flooding of the entire lake and reedbed area in 2050 and 2080. Under this management scenario, minimum lake water depths are projected to fall to depths of 50cm and 60cm, respectively, in the deepest areas of the lake in 2050 and 2080. This is in comparison to the *as is* and *fully open* scenarios, which produce minimum water depths of just 30cm in the deepest areas of the lake. At these minimum levels, in 2050 and 2080 up to 80% of the lake and 60% of the reedbed are exposed to <20cm depths, and water depths also fall to 0cm within the wetland under the *as is* and *fully open* scenarios.

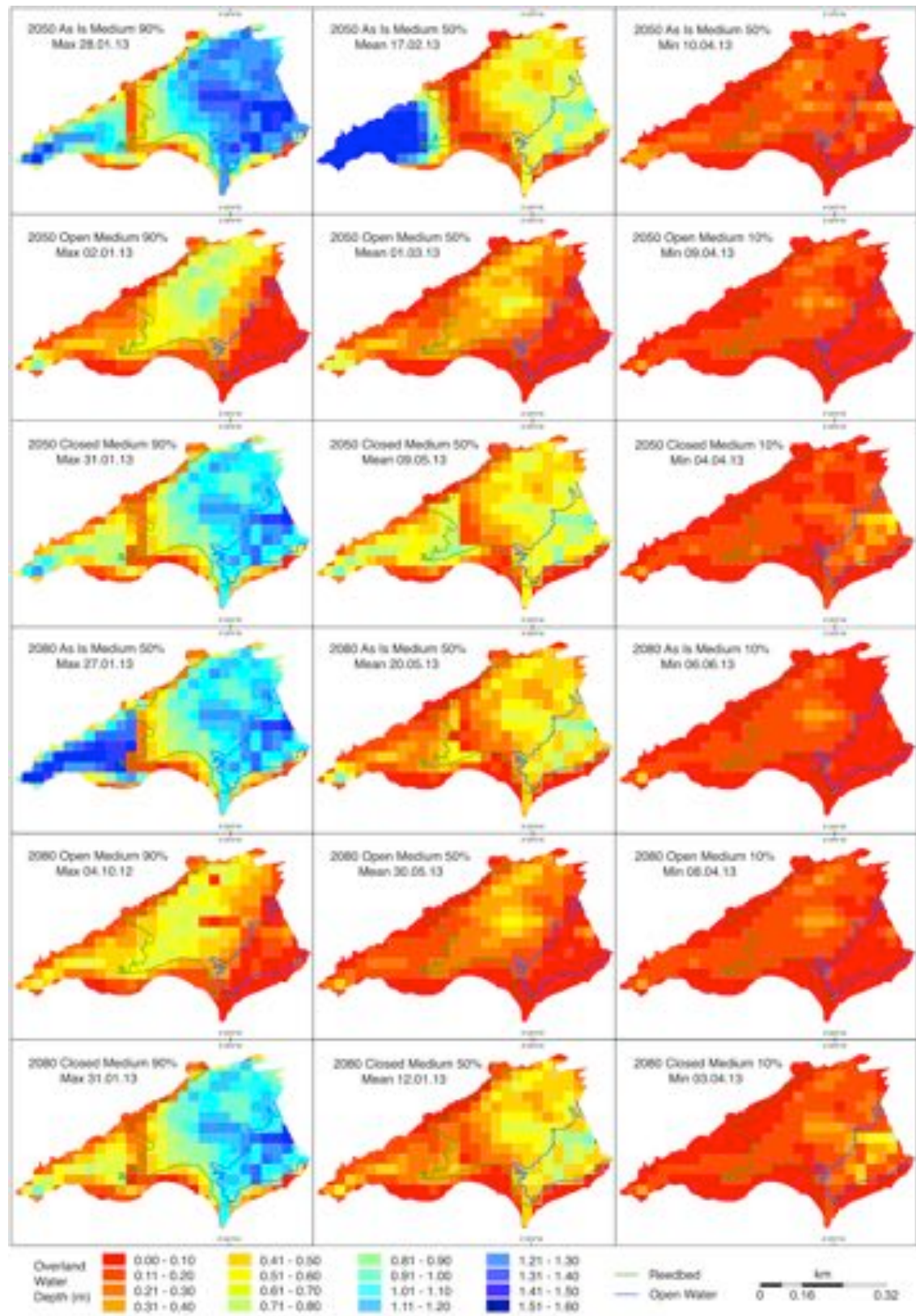


Figure 6.29 Impacts of the combined effect of future climate change and hydrological management on overland water depth across the Sheskinmore Lough lake and wetland system for a range of scenarios based on the maximum (Max), minimum (Min) and mean (Mean) modelled lake water levels for each management scenario (*as is*, *fully open* and *fully closed*) under the Medium emissions scenario for the 2050s and 2080s. Note the lake open water and reedbed habitats are outlined in blue and green, respectively, while the remainder of the modelled area encompasses the wetland habitat.

6.6 Discussion

In general, modelling results were in good agreement with observations. The 'flashy' nature of river discharge and the variable pattern in lake levels are clearly demonstrated, confirming the ability of MIKE SHE for effective modelling of the contemporary hydrology of the lake and wetland system at Sheskinmore Lough. The model was also able to simulate the impact of potential future scenarios associated with climate change and sluice management. This section discusses the likely impacts of these future hydrological conditions, both independently and combined, on the broader ecology of the site.

Acreman et al (2013) developed tools for assessing the responses of wetland hydrology to climate change and management to help conservation managers in the UK better understand likely future impacts on aquatic flora and fauna. Their approach to developing these assessment tools consisted of four elements including: 1) assessing future climate change scenarios and their associated uncertainties; 2) simulating potential catchment responses to these climate change scenarios; 3) exploring the likely plant community ecohydrological responses; and 4) identifying the tolerance threshold of ecosystems to climate change and management scenarios, i.e. whether a small or a large alteration in hydrological regime is needed to cause significant change. This thesis has already tackled elements 1) and 2) of this approach, therefore the remainder of this chapter sets out to tackle elements 3) and 4).

6.6.1 Ecological responses to climate change

Water levels across the lake and wetland system at Sheskinmore Lough are determined directly by precipitation onto the lake, evapotranspiration from it and upstream inputs, which themselves are impacted by the balance between precipitation and evapotranspiration, and outputs via the outflow channel. The future ecohydrology of the site will therefore be determined, both by changes in climate and its effects on fluxes to and from the lake, and also by modifications to riverine outflows via hydrological management, the latter of which will be discussed independently in the following section.

Precipitation and temperature (and in turn evapotranspiration) are predicted to change significantly with global climate change and are expected to directly impact coastal lake and wetland ecosystems such as Sheskinmore Lough over the coming decades (Anthony et al, 2009). In addition, indirect impacts such as changes to overland and groundwater flow are also important for the future sustainability of such habitats (Acreman & Blake, 2013). It should be noted, however, that great uncertainties exist in relation to the magnitude and extent of any projected hydrological changes in response to climate change (Thompson et al, 2009). The likely impacts of these changes on aquatic ecosystems are therefore also subject to the same uncertainties.

Hydrological modelling of the lake and wetland system at Sheskinmore Lough revealed that climate change has a noticeable impact on site water levels when contemporary hydrological management is unaltered. Specifically, mean annual precipitation is expected

to raise mean annual lake water levels by 0.03m (1%) in 2050 and by 0.04m (1%) in 2080 at the 50% probability level under the High emissions scenario. At this probability level, however, these increases are as likely as not to be exceeded, therefore the absolute maximum amount (90% probability level) that water levels are predicted to increase by is 0.08m (3%) in 2050 and 0.12m (4%) in 2080 under the High emissions scenario. Equally, the absolute minimum amount (10% probability level) that water levels are predicted to increase by is 0.01m (0.5%) in 2050 and 0.02m (1%) in 2080 under the High emissions scenario.

Simply observing the projected changes in mean annual water levels at all probability levels, however, masks the seasonal variation that was also apparent in the modelling results. Specifically, water levels in Sheskinmore Lough are projected to increase by an average of 37mm (2%) in 2050 and by 50mm (3%) in 2080 during autumn and winter, and are projected to decrease by an average of 37mm (2%) in 2050 and by 65mm (4%) in 2080 during spring and summer under the High emissions scenario. As these levels are projected at the 10% probability level, they are therefore very likely to be exceeded. Lake water levels are, however, unlikely to exceed increases of 4.5% in 2050 (H_{90}) or 5.5% in 2080 (H_{90}) in the autumn and 4.6% in 2050 (H_{90}) or 7.3% in 2080 (H_{90}) in the winter. Similarly, lake levels are unlikely to exceed increases of 0.6-0.8% in 2050 (H_{90}) or 1.7-3.1% in 2080 (H_{90}) in the spring and 1.3-1.7% in 2050 (H_{90}) or 1.4-1.8% in 2080 (H_{90}) in the summer. In addition, water levels are more than likely (10% and 30% probability levels) to decrease during the spring and summer (1.1% in 2050 and 2.3% in 2080) due to enhanced evapotranspiration caused by increased temperatures. Ultimately, the projections of increased autumn and winter water levels and potentially declining spring and summer water levels in Sheskinmore Lough show that intra-annual water level fluctuations are likely to increase over time.

Increased water levels during future autumn and winter periods, and slightly reduced water levels during future spring and summer months, will have a number of impacts on the ecology of Sheskinmore Lough. Lake biota respond differently to changes in water level, with studies recording just small changes in water level resulting in a large shifts in plant community composition (Coops et al, 2003; Mjelde et al, 2003). For example, littoral helophyte communities, including species such as *Equisetum fluviatile*, *Carex spp*, and *Phragmites australis*, found within and around Sheskinmore Lough, can be completely dependent on slight water level fluctuations to expose substrates for germination or flood seedlings (Wantzen et al, 2008; Mjelde et al, 2003). High water levels in winter may limit submersed plant expansion inducing a shift to a sparsely vegetated state, whereas a substantial reduction in spring lake level may encourage expansion of submersed plants. In the case of Sheskinmore Lough, spring water levels are likely to have an effect on macrophyte dispersal and expansion within and around the lake, especially community composition in the shallowest parts of the littoral zone.

Disturbance from intra-annual water level fluctuations can cause mortality of aquatic plants through heating and desiccation in summer and reduced light penetration due to enhanced inundation in winter (Blindow, 1992; Irwin & Noble 1996). Desiccation and

inundation tolerance determine the distribution of many littoral species including *Juncus bulbosus*, *Littorella uniflora* and *Equisetum fluviatile*, found along the aquatic-terrestrial transition zone of Sheskinmore Lough (Mjelde et al, 2003). Flooding of terrestrial wetland plants due to increased water levels around the lake fringe, when water depths across the system reach their maximum, has the contrasting impact of enhancing inundation, which in turn prevents respiration, reduces light levels required for photosynthesis, and initiates chemical changes (Middleton, 1995). The effects of these changes are consistent with disturbance factors that increase resource availability, cause the removal of dominant species and delay successional processes (Sousa, 1984). Disturbance can, however, also promote colonisation by a new suite of plant species (Salisbury, 1970; Mooij et al, 2005).

Indeed, a number of studies identified climate-induced intra-annual water level fluctuations as a key disturbance factor in terms of hydrological influence on littoral vegetation dynamics (Gill & Bradshaw, 1971; Nilsson, 1981; Gasith & Gafny, 1990; Irwin & Noble, 1996; Fraisse et al, 1997; Hroudova & Zakravsky, 1999; Keddy, 2000; Abrahams, 2008). According to Grime (1979), the key mechanism linking climate change-induced increased water level fluctuations with impacts on lake shoreline communities is likely to be the disturbance regime generated by repeated drawdown and re-flooding along lake shorelines. When combined with a fertility gradient, Grime's CSR theory proposes a three-way classification, dividing plant species into 'Competitive', 'Stress Tolerant' and 'Ruderal' groups depending on their observed traits (Abrahams, 2008). These classes are arranged on two opposing environmental axes, one describing the level of habitat disturbance, the other fertility (Figure 6.30). The two axes produce four possible types of environment, three of which are inhabitable by plants (Grime, 1979). Each environmental type corresponds to a vegetation group. For example, low disturbance-high fertile habitats favour competitive species, low disturbance-low fertility habitats favour stress-tolerant species, and high disturbance-high fertile habitats support ruderal species. The fourth type of environment, with high disturbance-low fertility, is uninhabitable (Grime, 1979).

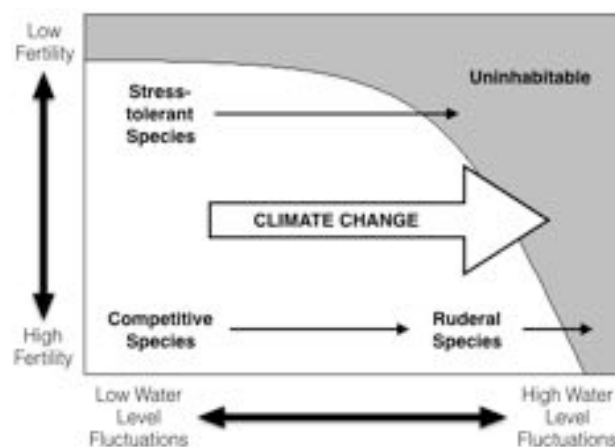


Figure 6.30 Grime's CSR triangle and climate change impacts on shoreline habitats with fertility and disturbance axes (Adapted from Abraham, 2008; p35).

Changes in species composition within Sheskinmore Lough shoreline habitats subject to increased water level fluctuations from climate change are likely to be dominated by a loss of competitive and stress-tolerant species with increasingly ruderal vegetation types and expanding areas of bare substrate. This will have significant impacts on the nature conservation value, ecosystem functioning and ecological services provided by the lake habitat. In a study of Great Lakes wetlands, Mortsch (1998) concluded that an increased frequency and duration of low water levels produced by climate change, together with changes in the timing and amplitude of seasonal water levels, would affect wildlife, waterfowl and fish habitats, water quality, wetland area and vegetation diversity. Other studies suggest, however, that lakes suffering a decline in biodiversity through artificial stabilization of water levels can experience a reversal of these adverse impacts (Wilcox & Meeker, 1991; Hill et al, 1998). They argue it is possible for climate change-driven fluctuations to restore biodiversity. Similarly, Abrahams (2008) suggests that lakes that have become dominated by extensive stands of large competitive species could, with increased water-level fluctuations, develop a higher species diversity through the creation of niches for less competitive species. What is clear is that the projected increases in intra-annual water level fluctuations in 2050 and 2080 under the three emissions scenarios are likely to be large enough to produce some level of heightened disturbance regimes in the future, especially along the lake fringe.

Whatever the resulting emissions scenario in 2050 and 2080, climate change is likely to have an impact on the ecology of Sheskinmore Lough, especially species and communities along the lake fringe and within the littoral zone. When combined with impacts from land management practices, for example, poaching by grazing of cattle around the lake fringe, the impacts of climate change are likely to be enhanced. Conservation management of the lake shoreline and littoral zone should therefore focus on four key areas: water level management, maximising favourable substrate conditions and shoreline topography, and encouraging vegetation establishment (Abrahams, 2005). These will allow the adverse impacts of increased disturbance to be mitigated and enhance fertility in these areas so that biodiversity can be maintained despite the effects of climate change.

6.6.2 Ecological responses to hydrological management

The projected increases in climate change-driven intra-annual water level fluctuations in 2050 and 2080 are likely to be exacerbated considerably by hydrological management in the future. Water level fluctuations, however, are natural patterns necessary for the survival of many species (Gafny et al, 1992; Gafny & Gasith, 1999; Wantzen et al, 2002; Wantzen et al, 2008). Although the majority of published research has focused on macrophytes, water level fluctuations are known to affect fish (Fischer & Ohl, 2005), macroinvertebrates (Grimas, 1961), waterfowl (McIntyre, 1994), and abiotic factors such as littoral nutrients, sediments and thermal stratification (Furey et al, 2004; Weston et al, 2004). Indeed, Gafny et al (1992), Gafny & Gasith (1999), Wantzen et al (2002) and Wantzen et al (2008) all observed that natural lake water level fluctuations maximise both biotic productivity and diversity. Indeed, Turner et al (2005) and Wagner & Falter (2002)

stress that lake level manipulations, specifically where water levels are kept constant for extended periods, result in decreased macrophyte diversity. Since the biota inhabiting lake systems such as Sheskinmore Lough have evolved with specific water level fluctuations, deviations from these patterns may exceed tolerance thresholds and dramatically alter community composition and diversity, especially in littoral areas (Sparks et al, 1998; Bond et al, 2008).

It is clear that hydrological management at Sheskinmore Lough to date has largely interrupted the natural intra-annual behaviour of the lake system. Modelling has revealed the sluice has two key impacts on water levels under current climate conditions. Firstly, when *fully closed* (or when closed during the *as is* scenario) the sluice maintains mean monthly baseline water levels at elevations between 2.84mOD and 3.23mOD (0.39m range). When *fully open* (or when open during the *as is* scenario) and under current climatic conditions, mean monthly water levels range between 2.60mOD and 2.70mOD (0.10m range). Although intra-annual water level fluctuations occur at these contrasting levels, they are eclipsed by the water level fluctuations imposed by the operation of the sluice during the *as is* scenario, which produces water level ranges of up to 0.85m. Secondly, the changes in water level resulting from open-closed sluice level shifts occur extremely rapidly. When the sluice is opened after a period of closure, rapid drawdown reduces water levels by up to 0.85m in less than 7 days. Equally, closure of the sluice following an open period leads to water levels rapidly rising by 0.85m in just 5 days.

Modified water level fluctuations pose lake systems with a number of potential impacts. Coops et al (2003) observed that shallow lakes in The Netherlands, in particular those in peatland habitats, have largely become degraded due to pollution, eutrophication and modified water level fluctuations. The latter are under strict control to prevent flooding, provide water supply for agriculture and maintain optimal navigation depth. The study, along with others (Ter Heerdt & Drost, 1994; Coops & Hosper, 2002), observed that impacts from modified lake water level fluctuations often result in deterioration of emergent vegetation along the shoreline, negative effects on the food chain and desiccation of adjoining wetland areas. With these impacts in mind, Coops et al (2003) acknowledged that climate change will likely accentuate these impacts and, in time, force a rethinking of water management. Indeed, they and a number of other studies observed that hydrologically manipulated lakes (including those like Sheskinmore Lough that are controlled by sluice structures) are more sensitive to climate-driven changes in hydrology than natural, open lakes (Street, 1980; Street-Perrott & Harrison, 1985; Bengtsson & Malm, 1997; Kebede et al, 2006).

Only since the late 1990s have aquatic ecologists begun to recognise the importance of water level fluctuations for temperate lake ecosystems (Hill et al, 1998; Wagner & Falter, 2002; Wilcox & Meeker, 1991; Wantzen et al, 2008). Hill et al (1998), Wagner & Falter (2002) and Wilcox & Meeker (1991) focused on the effect of intra-annual water level amplitude on macrophyte diversity via research geared towards evaluating the impacts of regulated water levels in reservoirs. These studies suggest that fluctuations between 1.5m and 2.0m are the optimal level for maximising macrophyte diversity. Similarly, a New

Zealand study focusing on the littoral macrophyte communities of 21 temperate lakes yielded similar results, suggesting that a 1.1m fluctuation resulted in the highest species richness in this climatic zone (Riis & Hawes, 2002). Conversely, the hydrology environmental standards for lakes proposed within the Water Framework Directive (WFD48) proposed a maximum permissible drawdown figure of 1m for macrophytes, which is based on annual lake level range data (Gosling & Hatton-Ellis, 2004). These conflicting figures indicate the maximum water level range that lake biota can withstand is fairly arbitrary. They also make no mention of the permissible rates of drawdown or rise in water levels. Such figures are therefore arguably largely dependent on lake morphology and should be used with caution, especially when applied to shallow lakes. For example, a 1m drawdown in a deep lake with steep sides would have little impact on the littoral habitat; however in a shallow lake with gradual marginal inclines the impact would be significant. Indeed, determining the optimum or threshold water level fluctuation range of a lake system, especially a shallow one, requires system-specific tailoring. In the case of Sheskinmore Lough, modelling has revealed that the current natural amplitude of water level fluctuations (i.e. when the sluice is *fully open* and when open during the *as is* scenario) is very small, at just 10cm. Sluice operation, however, increases this range by as much as 90cm. The shallow (<1.5m depth), flat (majority between 0.25m and 0.75m depth) morphology of Sheskinmore Lough means that sluice closure following a prolonged dry period (such as that observed from February to April in 2013), when water levels fall to their absolute minimum (to 30cm in the deepest areas), results in the open water area of the lake increasing by as much as 80% following sluice closure.

Large variations from natural lake level fluctuations have a number of ecological implications for Sheskinmore Lough. Firstly, raising water levels by up to 0.85m can noticeably alter the underwater light climate, especially in the deeper areas of the lake. Species that cannot withstand the lower light intensities associated with these increased depths are less likely to survive. Equally, they are more likely to be out-competed by species that thrive under lower light conditions. One such species is the rare submerged macrophyte *Najas flexilis*, which generally thrives at depths of up to 9m (Preston & Croft, 2001; Roden, 2002; Rostk & Schmidt, 2015). The species is rarely associated with depths below 1m, which dominate (85%) Sheskinmore Lough, therefore it is more likely to be compromised when the sluice is open during the summer months when water levels fall to their lowest levels. Secondly, altering the overall lake water level can dramatically change the spatial distribution of erosion and deposition zones (Wantzen et al, 2008).

Given the flat, shallow morphology of Sheskinmore Lough, the greatest impacts from erosion are most likely to occur at the lake fringe and across the flat shallow areas of the lake. When the sluice is open, lower water levels are likely to expose a greater area (up to 80%) of the lakebed to wind and wave erosion. Frequent high wind speeds associated with this area of Donegal, combined with the loose, soft sandy sediments that comprise the lakebed, means the majority of macrophytes within the shallow areas of Sheskinmore Lough are likely to be severely impacted when the sluice is open, especially during winter when storms are generally more intense. In addition, the rapid drawdown of water following the opening of the sluice is likely to enhance sediment erosion at the lake fringe,

as is the rapid raising of water levels following sluice closure. In areas of erosion, disturbance of lakebed sediments reduces the likelihood of germination and survival of macrophytes, especially those with shallow roots. Charophytes on the other hand do not possess roots and are fixed to the substrate via rhizoids (Hrivnak et al, 2001; Van den Berg et al, 2001; Van Nes et al, 2002). They are therefore more likely to thrive in these areas as they are able to be mobile with the sediment. Areas of deposition within the lake are likely to be concentrated around the lake fringe during periods of drawdown and rapidly rising water levels following sluice manipulation. When the sluice is open, however, lower water levels in the lake and the free movement of water downstream will mean that erosion is likely to be enhanced, and deposition decreased, especially at the mouths of the Duvoge and Abberachrin rivers and within and around the channel feature within the lake.

Thirdly, hydrological management appears to have an impact on *Phragmites australis* coverage. During the modelled survey period (18th June 2012 to 17th June 2013), the sluice was opened for an extended period from 22nd February to 9th April 2013 and was also open regularly following that initial period from 27th April to 17th May 2013, from 30th May to 25th June 2013, and from 10th July to 5th September 2013. Studies have shown that artificially lowering spring and summer water levels enhances the growth of *Phragmites australis*, which benefits from a competition-free environment during early summer when water depths are minimal (<30cm) (Van den Brink et al, 1995; Keto et al, 2002; Hellsten et al, 2006). In addition, Keto et al (2008) and Weisner (1987) discovered that regulation of Scandinavian lake water at low levels provides optimal growth areas for *Phragmites australis* as wave exposure enhances oxygen saturation within sediments.

Schmieder et al (2004) and Nechwatal et al (2008) both document the degradation of *Phragmites australis* belts along the fringe of Lake Constance, central Europe, following early spring floods when water levels are high. Extreme floods significantly reduce the oxygen supply to *Phragmites australis* rhizomes and submerged shoots, an impact that is becoming more widely accepted as a major factor in reed dieback (Koppitz, 2004; Ostendorp et al, 2003; Dienst et al, 2004), a phenomenon that is now under discussion in the context of climate change and other environmental changes on a regional scale (Keto et al, 2008). Opening the sluice for several extended periods during the modelling period, is therefore likely to have had a significant impact on the extent of the *Phragmites australis* reedbed. As submerged vegetation within Sheskinmore Lough is relatively sparse and competition in the littoral zone consequently low, this study would argue that low water levels are likely to favour the expansion of *Phragmites australis* stands. Indeed, the reedbed expansion identified by this thesis is therefore likely to be the direct result of lowered water levels from prolonged periods of sluice opening during the spring and summer months.

Variations in lake level fluctuations have a fourth potential ecological implication for Sheskinmore Lough. As well as their impact on vegetation within the lake, low spring and summer water levels will also have an impact on species of waterfowl, many of which depend on the Sheskinmore Lough and the wider wetland habitat for breeding and feeding. Lower spring and summer water levels, especially when the sluice is open, reduce

the area across the lake and wetland system suitable for waterfowl by approximately 80% (Etheridge et al, 2000). Etheridge et al (2000) describes the range of measures that have been used to improve conditions for breeding waders at freshwater sites which lack sufficient water levels during drier than average spring months. They increased water levels in spring and early summer using reservoirs to store water abstracted from rivers in winter when river flows are high. These reservoirs were designed with sufficient capacity and water supply to achieve optimum water levels across a wetland system in 75% of years under current climatic conditions, but with additional capacity to take account of projected future increases in water level. Measures were also taken to rotationally flood different areas instead of trying to keep a single large area of habitat flooded every year (Eglington et al, 2008). It was also important to ensure a suitable infrastructure was in place to remove excess water during extreme flood events, which are predicted to increase in the UK and Ireland in the future (Ausden, 2014). It should be noted, however, that extreme rainfall events can also cause spring and summer flooding, which reduces breeding productivity of waders (Green et al, 1987; Ratcliffe et al, 2005). The short periods when the sluice was closed in spring and summer during the hydrological monitoring period reveal that spring and summer water levels in the lake are raised to levels sufficient for breeding waders, but the lake fringe and wetland habitat may be vulnerable to excessive flooding during extreme rainfall events.

Ausden (2014) acknowledges that further reductions to water levels in lake systems during the spring and summer as a result of climate change may eventually make it impractical to maintain optimum hydrological conditions for breeding waterfowl. They therefore suggest it is acceptable to allow an increased rate of drawdown in water levels in spring and summer and encourage flooding in winter. This would mean that, in the case of breeding waders, areas might still remain wet enough during the breeding season despite a greater rate of drawdown. If applied at Sheskinmore Lough, this approach would centre around accommodating, rather than resisting changes in hydrology, specifically higher winter water levels and reduced spring and summer water levels, that occur as a result of climate change (Hoffmann, 1958; Valverde, 1958; Ausden, 2014). In order for this approach to work effectively, careful operation of the sluice would be required to ensure lake levels were not too low for too long during the spring and summer period. In addition, winter water levels should not be too high for too long. This approach would also have an important effect on the rare, protected whorl snail, *Vertigo geyeri*, which depends on summer groundwater levels within the wetland being maintained at or close to the ground surface (Cameron et al, 2003; Holyoak, 2005). Although section 4.4.2.3 shows there is no strong link between water levels in the lake and the western end of the wetland where *Vertigo geyeri* is most abundant (NPWS, 2006), significantly lower water levels within the lake in summer, combined with the eastward sloping topography, are likely to reduce groundwater levels across the whole site in a west-to-east direction.

Finally, low lake water levels during the spring and summer period when the sluice is open expose the vulnerable marginal and littoral habitats to poaching by grazing cattle. During the hydrological monitoring period cattle were frequently observed wading into the main body of the lake, primarily from its southern side, and on a couple of occasions

the cattle were observed crossing the lake along its shallow western side. At other times, including during the winter, signs of cattle poaching was evident in these areas, often reaching far into the lake by as much as 20m. In contrast, high water levels in the wetland (up to 30cm) enhanced by sluice closure are likely to increase poaching by grazing cattle and horses in the central and western arm of the wetland. Although Oliver (2007) acknowledges a certain amount of poaching disturbance can be beneficial for biodiversity and grazing of emergent vegetation is an important part of freshwater habitat management to control encroachment of marginal vegetation, too much poaching is likely to exceed the tolerance levels of the majority of aquatic macrophytes. Complete absence of grazing by livestock on the other hand is generally regarded as detrimental to the biodiversity of marginal aquatic ecosystems (Oliver, 2007).

As discussed in the previous section, moderate water level fluctuations make a positive contribution to the diversity and conservation value of shoreline vegetation (Smith et al, 1987; Pieczynska, 1990; Schneider, 1994; Hawes et al, 2003). The extreme fluctuations observed and projected under the *as is*, *fully open* and *fully closed* hydrological management scenarios, however, have revealed that hydrological management is likely to have more significant ecological impacts than climate change when the two factors are examined independently. What is less clear, due to inherent uncertainties in climate change scenarios, are how the potential impacts from hydrological management will be exacerbated or suppressed under future climate change. If water levels at Sheskinmore Lough have already reached an ecological threshold range under current conditions, then climate change has the potential to disrupt lake water levels to a point where the ecological tolerance threshold is exceeded. For instance, when combined with a fertility gradient, Grime's CSR theory (Figure 6.31) indicates fluctuations outside of this ecological threshold range should be limited when controlling lake water levels under climate change, especially for nutrient-poor systems such as Sheskinmore Lough (Abrahams, 2008).

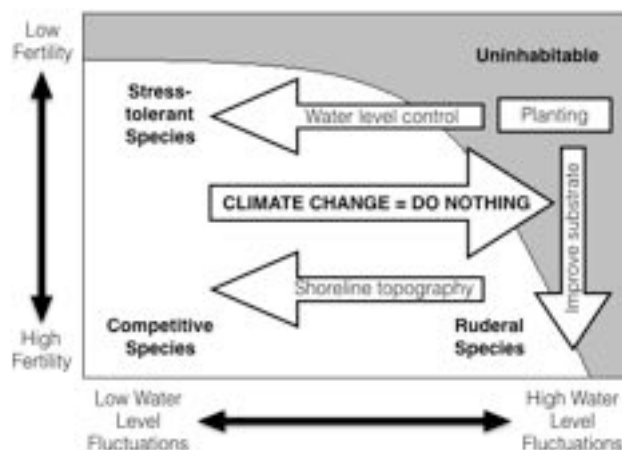


Figure 6.31 Grime's CSR triangle, climate change impacts on shoreline habitats with fertility and disturbance axes, and potential management options to allow adaption to climate change (Adapted from Abraham, 2008; p37).

6.6.3 Relative impacts of climate change and management on system ecohydrology

So far, this chapter has shown that climate change is likely to impact the ecohydrology of Sheskinmore Lough in the future, specifically: exploring the likely plant community ecohydrological responses; and identifying the tolerance threshold of ecosystems to climate change and management scenarios. Enhanced winter water levels, decreased summer water levels and greater intra-annual variation in the future are likely to affect the current lake ecology, especially in the littoral zone and marginal areas. Climate change is unlikely to be the only influencing factor on lake and wetland ecohydrology in the future, however, as hydrological management is predicted to continue to have a significant impact on the ecology of Sheskinmore Lough. The likely future ecohydrological impacts from the combined influence of climate change and hydrological management on Sheskinmore Lough and the surrounding wetland are summarised in Table 6.13. It should be noted that this summary is just a snapshot of potential future impacts at Sheskinmore Lough. The factors selected have been chosen because they are most reflective of this thesis as a whole. The summary does, however, go some way towards emphasising the complex array of potential influences that climate change and hydrological management, when combined, are likely to have on the future ecohydrology of the site. It also indicates the variety of flora and fauna, including vulnerable and prolific species that are dependent on the Sheskinmore lake and wetland system and are likely to be impacted. Hydrological managers therefore need to consider all of the potential impacts, the array of species, the specific ecohydrologies they depend on, as well as the overall biodiversity of the site when developing future management strategies.

The combined influence of climate change and hydrological management is being increasingly recognised as the most likely severe threat to aquatic flora and fauna globally (Quennerstedt, 1958; Grimas, 1961; Hellsten, 2001; Keto et al., 2006; Sutela & Vehanen, 2008). The most vulnerable communities within Sheskinmore Lough, those most susceptible to large variations in water level, are primarily found within the shallow littoral zone and at the lake margins. The range in water level fluctuations within the lake and their associated impacts are largely in these shallow and peripheral areas. Nevertheless, species dependent on the maintenance of high water levels in the deeper areas of the lake and across the wetland are also vulnerable. As climate change enhances precipitation in winter and reduces it in summer over the coming century, these impacts are likely to increase in magnitude over time.

Climate change is therefore likely to force hydrological managers in wet temperate regions, such as northwest Ireland, to adjust water level management schemes so that they are more sensitive and, ultimately, more adaptable to future changes. Future management should therefore allow a degree of water level fluctuation by combining ecosystem rehabilitation with hydrological functions (Coops & Hosper, 2002). To date, ecohydrological research has primarily focused on climate change prediction, impact assessment and mitigation, and there has been little attempt to develop practical adaptation methods to reduce potential climate change impacts on lakes and wetland systems (Hulme, 2005; Abrahams, 2008; Ausden, 2014). Such measures could increase the

flexibility of management of important sites such as Sheskinmore Lough, enhance the possibilities for complex ecosystems to adapt to change and reduce the additional pressures of non-climate related impacts (Abrahams, 2008; Ausden, 2014). A non-interventionist approach can be taken, accepting the changes to environments that will occur and allowing new habitats and communities to develop without substantial input. In many cases, however, a more appropriate approach is required, in order to implement active management strategies to ensure the most severe effects of climate change are mitigated, as long as they facilitate beneficial adaptation to altered hydrologic regimes (van Dam et al, 2002).

Table 6.13 Summary of the likely future ecohydrological impacts per season and sluice status from the combined influence of climate change and hydrological management at Sheskinmore Lough.

Sluice Status	Season	Climate Change and Hydrological Management Impacts
Open	Winter	Decreased water levels in the lake and wetland
		Lower magnitude and frequency of water level fluctuations
		Decreased risk of flooding
		Increased groundwater inputs to the lake
		Maintained macrophyte species diversity
		Reduced area for overwintering bird species
	Summer	Decreased water levels in the lake and wetland
		Exposed shoreline and littoral zone
		Increased groundwater inputs to the lake
		Increased macrophyte species diversity
		Expansion of <i>Phragmites australis</i> coverage
		Reduced <i>Najas flexilis</i> coverage
		Reduced viable wetland habitat for <i>Vertigo geyeri</i>
		Reduced breeding area for waterfowl
Closed	Winter	Increased water levels in the lake and wetland
		Lower magnitude and frequency of water level fluctuations
		Increased risk of flooding
		Decreased groundwater inputs to the lake
		Reduced macrophyte species diversity
		Enlarged area for overwintering bird species
	Summer	Increased water levels in the lake only
		Submerged shoreline and littoral zone
		Decreased groundwater inputs to the lake
		Maintained macrophyte species diversity
		Reduced <i>Phragmites australis</i> coverage
		Expansion of <i>Najas flexilis</i> coverage
		Increased viable wetland habitat for <i>Vertigo geyeri</i>
		Enlarged breeding area for waterfowl

6.7 Conclusions

The ability to predict future hydrological conditions and their potential impacts upon aquatic systems is vital if conservation management is to be successful and sustainable in the long-term (Brooks et al, 1991; Refsgaard et al, 1992; Jain et al, 1992; Thompson & Hollis, 1995; Lorup et al, 1998; Al-Khudhary et al, 1999; Karvonen et al, 1999; Christiaens & Feyen, 2001; Thompson et al, 2004; McMichael et al, 2006). On the whole, modelled projections were in good agreement with observations, confirming the ability of MIKE SHE for effective modelling of the contemporary hydrology of the lake and wetland system at Sheskinmore Lough. The model was also able to simulate the potential future hydrological conditions (specifically above-ground water levels and overland water depth) of the lake and wetland system under a series of climate change and management scenarios.

Modelling has revealed that hydrological management, specifically in the form of sluice operation, has had a dramatic effect on the ecohydrology of Sheskinmore Lough. When combined with climate change, however, these effects are even greater. Modelling has shown that climate change is predicted to increase the magnitude of changes with progressively higher emissions scenarios by the end of the current century. On the whole, simulated changes in mean monthly lake water level closely follow monthly projected changes in precipitation. From Low to High emissions scenarios there is a general positive trend towards higher projected mean water levels throughout the year in both 2050 and 2080, except during June, July and August when water levels are likely to decrease. There is, however, more uncertainty associated with projected water levels in 2080.

In order to help conservation managers at Sheskinmore Lough better understand the likely future impacts on aquatic flora and fauna, this chapter assessed the responses of the system hydrology to a series of climate change and management scenarios, the latter of which was examined under both current and future climate. Impacts were assessed in the context of potential catchment responses to changing water levels, specifically via an exploration of the likely plant community ecohydrological responses and identification of the tolerance threshold of the lake and wetland system to climate change and management scenarios.

Precipitation is expected to raise mean annual water levels in the lake during the current century under all emissions scenarios. The largest increases, however, are predicted to occur under the most unlikely, or uncertain probability levels. Seasonal variation was also predicted to occur. Specifically, water levels in Sheskinmore Lough are projected to increase considerably in the autumn and winter, but are expected to increase only slightly in the spring and summer, and may decrease during the spring and summer in 2080 due to enhanced evapotranspiration driven by increased temperatures. Ultimately, projections of increased winter water levels and declining summer water levels indicate that intra-annual, or seasonal, water level fluctuations are likely to increase over time.

This is likely to have a number of impacts on the ecology of Sheskinmore Lough. Even small changes in water level can result in large shifts in plant community composition, within shallow freshwater aquatic systems due to factors such as disturbance, heating,

desiccation and inundation, while some species can be completely dependent on specific water level fluctuation patterns. At Sheskinmore Lough, low spring water levels are likely to have an effect on macrophyte dispersal and expansion within and around the lake, especially community composition in the shallowest parts of the littoral zone. Indeed, climate-induced intra-annual water level fluctuations are likely to be key disturbance factors in terms of hydrological influence on littoral zone vegetation dynamics.

When combined with a fertility gradient, the disturbance regime generated by repeated drawdown and re-flooding along the lake shoreline is likely to be exacerbated, and therefore has the potential to exceed ecological tolerance thresholds. These fluctuations are therefore likely to turn habitable areas of the lake into those that are largely uninhabitable by existing species, and thus dramatically influencing community composition. This will have significant impacts on the nature conservation value, ecosystem functioning and ecological services provided by the lake habitat. Whatever the resulting emissions scenario, climate change is predicted to impact the ecology of Sheskinmore Lough, especially along the lake shoreline and within the littoral zone.

In addition to climate change, three management scenarios were assessed based on the three possible decisions that could be made by water managers at the site: the sluice varied *as is* (i.e. opened and closed) in response to changing water levels; the sluice left *fully open* at all times; and the sluice left *fully closed* at all times. The projected increases in climate change-induced intra-annual water level fluctuations in 2050 and 2080 are likely to be exacerbated by hydrological management in the future. Variations in water level are far greater under the *as is* scenario in comparison to the *fully open* and the *fully closed* scenarios. The act of manipulating the sluice from closed to open and then closed again results in water levels falling by 0.85m in less than 7 days and subsequently rising by 0.85m in just 5 days. The open water habitat area of the lake is therefore significantly impacted by hydrological management, decreasing substantially in area and exposing up to 80% of the lakebed to water depths below 30cm.

Large lake level fluctuations such as these have a number of ecological implications for Sheskinmore Lough including: reducing light availability in deeper areas; redistributing erosion and deposition zones and associated disturbance; increasing *Phragmites australis* coverage during summer months; reducing suitable habitat areas available to breeding waterfowl and the rare whorl snail *Vertigo geyeri*. In addition, raising water levels by up to 0.85m can noticeably alter the underwater light climate, especially in the deeper areas of the lake. Species that cannot withstand the lower light intensities are likely to be compromised. One such species that is actually likely to benefit from raised summer water levels, however, is the rare submerged macrophyte *Najas flexilis*, which generally thrives at depths of up to 9m.

Water level fluctuations are, however, necessary for the survival of many species and can maximise both biotic productivity and diversity. Since the biota inhabiting Sheskinmore Lough have evolved with specific water level fluctuations, deviations from these patterns may exceed tolerance thresholds and in turn dramatically alter community composition and diversity. Hydrological management at Sheskinmore Lough has largely interrupted

the natural intra-annual behaviour of the lake system. Modelling has revealed the sluice has two key effects: generating artificially large ranges in water levels and rapid drawdown and inundation rates, that have the potential to threaten lake systems with a number of negative impacts. These findings are particularly concerning given the current dearth of knowledge regarding lake level fluctuations and apparent conflicting opinions over the maximum lake water level range that aquatic biota can withstand. Ultimately, determining the optimum, or threshold, water level fluctuation ranges of a lake system, especially such a shallow system as Sheskinmore Lough, requires system-specific tailoring. In the case of the present system, modelling has revealed the natural amplitude of water level fluctuations is very small (10cm). The shallow, flat morphology of Sheskinmore Lough means that sluice closure, following a prolonged dry period, results in the open water area of the lake increasing by as much as 80%.

The extreme fluctuations observed and projected under sluice operation have revealed that hydrological management has a more significant ecological impact on site biota than climate change, when the two factors are examined independently. When combined, however, impacts from both of these anthropogenic forces on the ecohydrology of the site together present the greatest overall threat. This study has acknowledged that inherent uncertainties attached to climate change scenarios mean it is unclear exactly how these impacts will change in the future and exactly when climate change is likely to exceed the ecological threshold of the lake and wetland system. What is clear, however, is the complex array of potential impacts that climate change and hydrological management, when combined, are likely to have on the future ecohydrology of the site. The variety of biota under threat in the future means hydrological managers need to consider all of the potential impacts, the array of vulnerable and invasive species, the specific ecohydrologies they depend on, as well as the overall biodiversity of the site, when developing future management strategies. From birds to snails and from rare plants to prolific invaders, a delicate balance, between hydrological management for the maximisation of current biodiversity and hydrological management to enhance ecological resilience, must be achieved going forwards.

7 Conclusions and recommendations

7.1 Introduction

The results of this study have revealed a lake and wetland system that has a complex ecohydrology set in a complicated coastal environment. The position of the lake and wetland system within Sheskinmore Lough SPA, spanning a transition from calcareous dunes to acidic peat overlaying solid siliceous bedrock, provides one of the clues as to why the site encompasses a mosaic of different aquatic environments. Regional and local climatology, characterised by high rainfall, strong winds and low temperatures, also adds to this complexity. In addition, the varied topography of the site, which consists of high peat uplands to the north and west and undulating dunes to the south and east of the very flat plain occupied by the lake and wetland, promotes contrasting hydrological settings dominated by surface water and ground water processes, respectively. To add a further dimension to this complexity, the installation and *ad hoc* operation of a sluice within the outflow of the lake, to control water levels for conservation purposes, has introduced significant ecohydrological pressures across a number of the lake and wetland habitats.

This study recognises that, in order for conservation of Sheskinmore Lough, or any other shallow lake and wetland system, to be effective in the long-term, it is vital to develop a better understanding of the contemporary and past ecohydrology in order to determine the relative controls of environmental change and anthropogenic intervention. Through an assessment of the contemporary ecohydrological conditions of the lake and wetland system, this study has provided a suite of scientific evidence and understanding necessary for the formulation of robust historical reconstructions and reliable predictions of the likely impacts of future climate change and hydrological management.

7.2 The ecohydrology of Sheskinmore Lough: past, present and future

The coastal Atlantic location of Sheskinmore Lough means the lake and wetland system is exposed to low mean annual temperatures, high mean annual wind speeds (often gale force), and high mean annual rainfall. The study period (June 2012 to June 2014) was characterised by average temperatures and above average annual precipitation when compared to regional data since 1957. Regular large precipitation events, most significant during the autumn and winter months, were interrupted by a number of extended (>1 month in duration) dry periods during the spring and summer months.

The contemporary wetland system is typical of an open water transition fen, comprising species and communities associated with fen and mire habitats and oligotrophic hydrologies with circumneutral pH levels, low conductivities, and peaty or sandy calcareous substrates. The present lake system is typical of small, shallow, calcareous, lowland lakes found throughout the UK and Ireland, and its species and communities are also typical of the complex suite of environmental conditions that characterise the site. Within the wetland, hydrological inputs and topographic variation largely determine the distribution of terrestrial macrophyte communities. At the western, or terminal, end of the

wetland where groundwater influence is significant, species diversity is greatest. In contrast, the lowest diversity was found closer to the lake at the lower elevations in the centre of the wetland where groundwater flow is primarily influenced by the hydrology of the lake. Disturbance from grazing and land management at the western end of the wetland may also play a part in enhancing community diversity in that area. Within the lake, the highest community diversities are found in areas of moderate depth, especially on the soft sediments of the sloping channel margins, and in the areas near the Abberachrin and Duvoge river inlets. Littoral communities, including the rare *Najas flexilis*, thrive on the organic silt substrates at the base of the channel, while charophytes dominate the sandy substrates in the shallow zone where diversity is consequently extremely low.

The present lake system is characterised by a variable, primarily surface water-dominated hydrology. Flashy responses to precipitation events within the inflowing rivers are mirrored, albeit to a lesser magnitude, within Sheskinmore Lough, Sandfield Lough and the various beach and dune ponds indicating that the aquatic water bodies are all responsive to direct precipitation. In contrast, the wetland system is characterised by a more consistent hydrology dominated by groundwater processes. The installation and intermittent operation of a sluice in the outflow of the lake further accentuates the sporadic hydrology within the lake system, which in turn influences the groundwater levels across some parts of the wetland system.

Most notably, this study has identified that the operation of the sluice causes water levels within the lake to fluctuate by up to 1m. Sluice closure results in mean annual water levels increasing by as much as 0.85m within 7 days, while sluice opening causes water levels to fall by as much as 0.85m in just 5 days. The knock on effects of these significant lake level fluctuations were observed at hydrological monitoring sites surrounding the lake, as well as at those located a further distance west of the lake and reedbed within the central wetland area. Only the western end of the wetland and surrounding dune wetlands appear to be unaffected. In addition, the impacts from this hydrological disturbance are also reflected in the contemporary site ecology, especially the distribution of macrophyte communities, the abundance of *Chara aspera* in the lake, the endangered status of *Najas flexilis*, and the encroachment of *Phragmites australis* across the wetland.

Multiproxy paleolimnological analysis combined with examination of historical maps, charts and aerial photographs of Sheskinmore Lough indicate that hydrological variability is not confined to the contemporary era. Although the estuarine to freshwater transition was not captured in this study, analysis of lake cores revealed two important climatic and hydro-geomorphological shifts that define three key phases in the recent ecohydrological history of the lake and wetland system. Prior to the mid-1500s, the site was dominated by sandy conditions with low diatom and macrofossil abundance and diversity, but transitioned in the late 1500s to one dominated by peat, which persisted to the early 1800s. During this first phase, the environment would have been riverine, and conditions drier and windier than average, which led to low productivity at the site. A changing climate toward the end of the 1500s brought colder and wetter conditions associated with the Little Ice Age, which encouraged development of primarily riverine, peat-dominated

conditions at Sheskinmore. The ecohydrology of the system during this period is characterised by a surge in diatom and macrofossil abundance and diversity, the latter of which is dominated by *Phragmites australis*.

The decline in *Phragmites australis* and the appearance of Sheskinmore Lough as a lake feature in the historical map from 1835 implies that the transition from a riverine to lake system occurred in the early 1800s. It is likely that this shift was linked to increased flooding during wetter than average climatic conditions, but also possible that sand dune dynamics blocked the river course to force lake development. This most recent phase, covering the last 200 years, is characterised by a fluctuating aquatic environment that incorporates periods of peat or soil development, interrupted by layers of aeolian sand deposition. The ecohydrology of the system is characterised by shifts in diatom species dominance and reduced macrofossil abundance; however the macrofossil richness remains relatively consistent as a proportion of the lake system develops into terrestrial wetland. In particular, the wet and stormy conditions of the most recent period promoted successive periods of aeolian sand deposition across the site. Water level fluctuations are evident in the paleoecology of this phase implying that there have been periods of wetland encroachment and periods of flooding.

Since its formation, the ecohydrology of the lake has ranged from a riverine system that would have been dominated by emergent vegetation, to one that is now dominated by submerged macrophytes and charophytes. The diatoms and macrofossils in the sediment record reveal the site has been under the influence of oligotrophic freshwater since the mid 1500s AD, with no brackish or marine influence evident, other than from material transported to the lake and wetland from the adjacent coast by aeolian processes. The hydrochemistry of the system has fluctuated around circumneutral pH, ranging between slightly acidic to slightly alkaline conditions in response to periods of peat development and aeolian sand deposition, respectively.

Prior to sluice installation, fluctuations in lake water level would have largely been driven by a changing climate that forced shifts in both levels of precipitation and wind-driven sediment transport. This study would argue, however, that installation of the sluice has interrupted that natural pattern of hydrological variation. For instance, when the sluice is open, water levels in the lake fall considerably and do not recover following high rainfall events. Indeed, when the sluice is open, mean monthly water levels vary by just a few centimetres in contrast to fluctuations of magnitude closer to a metre when the sluice is closed. It is therefore likely that, if the sluice were left permanently open and under current climatic conditions, after a couple of years the lake system would experience encroachment from marginal wetland communities. If so, the open water system would reduce in extent, with possible transition back to a channelled, riverine system fringed by reedbed and an extensive terrestrial wetland habitat. Although operation of the sluice is raising water levels away from their natural state under current climate conditions, it does ensure a larger open water area is maintained.

Hydrological modelling revealed that climate change has a noticeable impact on site water levels when patterns of observed contemporary hydrological management (i.e. sluice operation) are unaltered. Although there are large uncertainties attached to these

projections, the future climate is generally expected to be wetter in this region of northwest Ireland, especially during the winter months. In addition, evapotranspiration is most likely to increase all year round. Evapotranspiration will be especially prevalent during the windiest months in winter and early spring and is expected to offset precipitation at the site, albeit only slightly as mean monthly temperatures will still remain low. Precipitation is expected to raise mean annual water levels in the lake during the current century under all possible emissions scenarios. Unsurprisingly, the largest increases in water levels are predicted to occur at the most unlikely, or uncertain, probability levels. The increase in mean annual water levels, however, masks a pattern of intra-annual variation. Specifically, water levels in Sheskinmore Lough are projected to increase considerably in the autumn and winter, and are expected to decrease slightly in the spring and summer under the majority of scenarios and probability levels. The overall range of intra-annual water level fluctuations is therefore likely to increase in magnitude over time.

Changes in the pattern of water level fluctuation are likely to have a number of impacts on the site ecology. Most notably, large variations in intra-annual water levels have the potential to cause large shifts in plant community composition within the lake due to factors such as disturbance, heating, desiccation and inundation. In particular, climate-induced intra-annual water level fluctuations are likely to be a key disturbance factor in the littoral zone and marginal areas of the lake. When combined with a fertility gradient, the disturbance regime generated by repeated drawdown and re-flooding along lake shorelines has the potential to create environments that are inhabitable by plants. Changes in species composition within Sheskinmore Lough shoreline habitats subject to increased water level fluctuations from climate change are therefore likely to be dominated by a loss of competitive and stress-tolerant species with increasingly ruderal vegetation types and expanding areas of bare substrate.

The projected increases in climate-driven intra-annual water level ranges over the current century are also likely to be significantly influenced by the sluice. Although a certain degree of water level fluctuation is necessary for the survival of species and maximisation of biodiversity, deviations from current patterns of hydrological management may exceed ecological tolerance thresholds and dramatically alter macrophyte community composition and diversity. Climate change is likely to increase water level ranges beyond natural tolerance levels and thereby exacerbate the impacts of the sluice.

Analysis of contemporary hydrological data prior to and during modelling revealed the sluice has two key impacts. First, water level fluctuations imposed by the operation of the sluice greatly exceed natural water level fluctuations. Second, this range of sluice-driven water levels is generated extremely rapidly following sluice adjustment. Large, rapid lake level fluctuations have a number of ecological implications for Sheskinmore Lough including: reducing light availability in deeper areas; redistributing erosion and deposition zones and associated disturbance; increasing *Phragmites australis* coverage during summer months; and reducing suitable habitat areas available to breeding waterfowl and the rare whorl snail *Vertigo geyeri*. In addition, water level increases of up to 1m in a lake that does not exceed 1.5m in depth has the potential to noticeably alter the underwater

light climate, especially in the deeper areas of the lake. Species that cannot withstand the lower light intensities are likely to be compromised. One such species that is likely to benefit from raised summer water levels, however, is the rare submerged macrophyte *Najas flexilis*, which generally thrives at depths of up to 9m. When examining the past environment of the lake, this study discovered that water level fluctuations of similar magnitude to those created by the sluice are likely to have occurred over the last couple of centuries. In addition, paleoecological analysis indicated that biota within the lake have clearly managed to adapt to these fluctuations in the past. The frequency of these fluctuations, however, was on the interannual, rather than intra-annual scale and therefore would have had different, and possibly more gradual, impacts on lake and wetland ecosystems.

The large water level fluctuations observed and projected under sluice-driven hydrological scenarios in this study have therefore revealed that hydrological management has a more significant ecological impact than climate change when the two factors are examined independently. However, it is the combined effects of these two drivers of change that will have the greatest impacts on the ecohydrology of Sheskinmore Lough. Due to large uncertainties associated with climate change, the magnitude of potential future impacts is unclear. Nevertheless, this study would emphasise that it is very possible water levels at Sheskinmore Lough have already reached an ecological tolerance threshold range under current management conditions. If so, by the end of the century, climate change has the potential to raise water levels above this tolerance threshold and thereby accentuate impacts already imposed by the sluice, which will generate a whole host of challenges for conservation managers.

7.3 Contributions to coastal freshwater science

This thesis intended to improve knowledge of the current ecohydrology of Sheskinmore Lough SPA, but to also enhance the wider understanding of shallow coastal lake and wetland ecosystems as research on their formation, development, ecohydrology and environmental functioning is limited (Van Groenendael et al, 1993), and in Ireland is negligible (Caffrey et al, 1999). Prior to the Habitats Directive, only four coastal lakes in Ireland had received noteworthy research attention (Oliver, 2007). Indeed, coastal lagoons are the only coastal lake type that has been extensively studied in Ireland (Good & Butler, 1998; Hatch & Healy, 1998; Healy & Oliver, 1998; Oliver & Healy, 1998; Healy, 1999a; Healey, 1999b; Healey, 1997; Roden, 1999; Good & Butler, 2000). A survey of coastal lagoons in the Republic of Ireland between 1996 and 1998 was conducted by the National Parks and Wildlife Service (NPWS) in response to the Habitats Directive, to reduce the knowledge gap and facilitate the selection of protected sites for designation. The majority of systems surveyed however, were marine or brackish, and the number of studies focusing on freshwater coastal lakes in Ireland is extremely small. The lack of research on coastal freshwater lake systems may be due to the fact that they are highly dynamic, transitional ecosystems, constantly evolving with the coastal framework. Environmental conditions within coastal lakes vary considerably as a result (Van Groenendael et al, 1993). Salinity, pH, temperature and turbidity, all fluctuate both on a

spatial and a temporal basis.

Coastal lake ecosystems are especially difficult to study when it comes to measuring ecosystem condition over a short timeframe. The studies that do exist primarily focus on answering specific questions using focussed approaches. For example, Holmes et al (2007) used paleolimnological techniques to identify Holocene coastal lake level and salinity changes, whilst Woodward et al (2012) conducted statistical analyses to determine environmental drivers of coastal lake surface sediment chemistry. Studies of coastal lakes in other parts of the world show similar focused aims and methodologies including: developing an integrated model framework in order to assess coastal lake sustainability in Australia (Ticehurst et al, 2007); using multi-proxy approaches to determine vegetational fluctuations within coastal lakes in Italy (Rita et al, 2010); and conducting ecological surveys to investigate factors affecting the distribution of coastal lake macrophytes in Estonia (Karus & Feldmann, 2013). Given the dearth of research to date, this thesis offers a significant contribution to an otherwise under-researched topic. Not only is the study unique in its exploration of the interaction between a coastal lake system and its adjoining habitats, the use of a multidisciplinary approach has delivered a comprehensive baseline and integrated understanding of the ecohydrology of shallow lake-wetland systems.

7.4 Recommendations

7.4.1 Conservation management

Through improved understanding of the likely future impacts of factors such as climate change and water management on the ecohydrology of shallow lake-wetland systems, this thesis can deliver some clear perspectives on future conservation management. The research presented here has established that the combined effect of climate change and hydrological management is likely to have the most significant impact on the future ecohydrology of Sheskinmore Lough. The variety of flora and fauna dependent on the lake and wetland system means hydrological managers need to consider all of the potential impacts these future influences may bring. As well as the overall biodiversity of the site, the ecohydrological requirements of an array of vulnerable species require particular attention when developing future management strategies. Overall, however, the focus should be on maintaining the delicate balance between hydrological management to sustain and, ideally, maximise biodiversity, and also to enhance ecological resilience to climate change. It is the success or failure of these strategies that will determine the nature conservation value, ecosystem functioning and ecological services provided by the lake-wetland habitat in the future.

Modified water level fluctuations are becoming an increasing threat to lake systems. This is especially concerning when considering aquatic ecologists are only just beginning to recognise the importance of water level fluctuations for temperate lake ecosystem biodiversity (Hill et al, 1998; Wagner & Falter, 2002; Wilcox & Meeker, 1991; Wantzen et al, 2008). In addition, there are conflicting opinions over the maximum lake water level ranges that aquatic biota can withstand, as determining the optimum water level

fluctuation range of a lake system, especially a shallow one, usually requires system-specific tailoring (Gosling & Hatton-Ellis, 2004). In the case of Sheskinmore Lough, modelling has revealed that the natural amplitude of water level fluctuations is very small. Sluice operation, however, increases this range dramatically. The shallow, flat morphology of Sheskinmore Lough means sluice closure following a prolonged dry period results in the open water area of the lake increasing considerably. This thesis therefore recommends that future conservation management should primarily focus on minimising the extreme water level fluctuations imposed by the sluice.

This study recommends that minimising extreme water level fluctuations should follow two distinct sets of guidelines. The first are based on current ecohydrological conditions observed within the lake and across the wetland system and should involve limiting sluice openings to occasions following very high rainfall events only. At certain times of year, for example in spring and summer, it may be beneficial to closely monitor water levels following sluice openings so that the sluice can then be closed at the appropriate time, for example as soon as water levels have dropped to moderate rather than low levels. On occasion during the spring and summer, however, it may be advantageous to allow water levels to fall to their lowest level as a certain level of disturbance is required to enhance biodiversity. This study would stress, however, that water levels should not be left too low for too long during the spring and summer months if lake biodiversity and populations of species such as the rare *Najas flexilis*, that prefer greater water depths, are to be maximised. In addition, prolonged low water levels during the summer period is likely to encourage further expansion of the *Phragmites australis* reedbed, both within the lake and from its margins within the wetland system. Species such as the rare whorl snail *Vertigo geyeri* and macrophyte communities identified in the wetland system, dependent on maintenance of groundwater levels during the summer, will also benefit from reductions in the time periods that the sluice is left open during these months.

As climate changes over the course of the current century, hydrological management will need to adapt accordingly. The second set of guidelines are therefore based on a longer term perspective and require strategic planning to put in place additional measures with the aim of maximising the resilience of the site to the potential additional pressures from climate change. This approach suggests modifying the sluice structure so that lake water can be controlled at different levels within the current two hydrological extremes determined by opening and closing. In addition, this study would recommend installing equipment to allow the lake levels to be monitored remotely. This could be in the form of a water level sensor attached to the sluice structure, or more simply, in the form of a video camera and stage board. Although each of these measures involves a certain amount of investment, remote monitoring of water levels would be the most proactive approach to obtain the necessary evidence with which to inform sluice decisions. If no action is taken to more sensitively control water levels at Sheskinmore Lough over the next couple of decades, then climate change is likely to exceed the current water level fluctuation threshold, thereby severely compromising the resilience and biodiversity of the lake and wetland system over the long-term.

Ultimately, the findings of this thesis have, not only enhanced existing site knowledge, they

have also contributed to wider scientific understanding of contemporary coastal freshwater lakes and wetland systems necessary to inform conservation from local to international level. This enhanced understanding will help ensure future conservation management strategies are underpinned by strong scientific foundations, with a focus on maximising resilience, preserving diversity, and enhancing the natural and sustainable functioning of coastal freshwater lake and wetland systems.

7.4.2 Further research

This multidisciplinary thesis has produced a comprehensive account of the past, present and future ecohydrology of Sheskinmore Lough. Nevertheless, there are a number of aspects of the research that should be developed further if conservation management of the species, communities and environment central to this study is to be effective over the long-term. The suggested areas of development are summarised in Table 7.1. Areas, which this study believes require the most urgent attention, are underlined. For example, within the lake, specific analysis of the year-to-year ecology of *Najas flexilis* should be conducted as soon as possible as the varied inter-annual abundance that this study observed indicates the species may be succumbing to unseen pressures that are temporally inconsistent. In addition, investigations should be carried out within the lake to determine whether aquatic macrophyte communities have reached an ecological threshold under current hydrological management. This is essential if managers are to develop appropriate effective hydrological management strategies in the future.

Table 7.1 Suggested areas of development for further research on species, communities and environment of selected habitats at Sheskinmore Lough SPA.

Habitat	Species	Communities	Environment
Lake	<u>Investigate the interannual ecology of <i>Najas flexilis</i> in comparison to other lake systems to better understand its environmental tolerance range</u> Establish the influence of water levels on select species at opposite ends of the water level spectrum	Examine the interannual ecology of the lake macrophyte communities to see how they vary year on year <u>Determine whether aquatic macrophyte communities have reached an ecological threshold under current hydrological management</u>	Comparison of the Sheskinmore Lough limnological environment with surrounding lakes and coastal lakes further afield Investigate the long-term ecohydrological history of the site prior to the early 1500s
Wetland	A detailed survey of groundwater hydrology to determine the impacts of the sluice on <i>Vertigo geyeri</i>	Investigate land management practices to establish their relative impacts on wetland communities	Develop an understanding of the impact land use practices have had on the wetland environment
Reedbed	Monitor the expansion of <i>Phragmites australis</i> to determine its potential future impact on the lake and wetland	Conduct a full survey of the reedbed to understand how water levels impact its communities	Determine the impact of the reedbed on the hydrogeomorphology of the lake and wetland
Dunes	Establish whether <i>Najas flexilis</i> or other rare species are present in the dune ponds and, if so, determine their significance	Survey the ecology of the dune ponds with Sheskinmore Lough to identify vulnerable species and communities	Assess the impact of aeolian-driven sand deposition on the lake, wetland and surrounding habitats

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Appendix

Appendix A. Wetland and reedbed plant communities identified during the June 2012, 2013 and July 2014 surveys assessed through TWINSpan, their associated species, National Vegetation Classification (NVC) and Natura 2000 'EUR27' codes (Elkington et al, 2001; EC, 2007).

Community	NVC, EUR27	Indicator Species	Description
1. Fen Meadow	M22, 21A0	<i>Plantago lanceolata</i> , <i>Ranunculus sceleratus</i> , <i>Trifolium pratense</i>	<p>Lowland first-level fen-meadow community characteristic of moist, base-rich and moderately mesotrophic peats. This community comprises secondary herbaceous vegetation found either in, or around, well-developed springs, flushes and mires, or marking out areas of surface or groundwater influence. It marks out soils that are moist for most of the year and have a moderate to high base-status, and usually a pH range of 6.5-7.5. It also tends to be dependent on maintenance via light mowing and grazing, otherwise bulky dominants and ranker grasses expand and overwhelm smaller herbs.</p> <p>Rushes and sedges of moderate stature are important, whilst grazing maintains a high density of associated flora. In summer this rush and sedge layer can be overtopped by flowering dicotyledons, especially <i>Filipendula ulmaria</i>, <i>Trifolium pratense</i> and <i>T. repens</i>, as well as <i>Ranunculus sceleratus</i>, which is often associated with areas of cattle poaching. Grasses are moderately abundant and include <i>Anthoxanthum odoratum</i> and <i>Poa flexuosa</i>. The presence of <i>Briza media</i>, <i>Leucanthemum vulgare</i>, <i>Lotus corniculatus</i>, <i>Plantago lanceolata</i> and <i>Potentilla erecta</i>, however, indicate stabilised, base-poor soils more characteristic of calcareous sand dunes. Moderate influence from reed bed encroachment is also evident by the presence of <i>Phragmites australis</i>.</p>
2. Peat Mire	M9, 7140	<i>Cardamine pratensis</i> , <i>Equisetum fluviatile</i> , <i>Sphagnum</i> sp.	<p>Lowland first-level mire community characteristic of soft, spongy peats kept permanently moist by moderately base-rich and calcareous waters. Waters and substrates always have a pH of 5 and above. This community is commonest in wetter areas, often located in natural hollows or old peat-workings, but also around springs in areas protected from grazing. This community is indicative of both surface water-fed topogenous and groundwater-fed soligenous mires. It cannot tolerate any other than very light or sporadic grazing, although it is normally too wet to be grazed but in some areas it occurs within mowing marsh that is periodically cropped. For example, this community can be found in wet field bottoms and edges that have been fenced off, and beside ditches and pasture boundaries. Progression to woodland, even in the absence of treatments such as grazing or mowing, appears to be slow. Under certain conditions, the community may develop to poor-fen and ombrogenous mire through the local formation of <i>Sphagnum</i> nuclei.</p> <p>This community has a diverse composition and physiognomy, even within individual stands, but is generally characterised by a fairly rich assemblage of sedges and vascular plants over a carpet of bulky, yet localised, <i>Sphagnum</i> sp. patches. <i>Menyanthes trifoliata</i> is common and often forms floating rafts. <i>Eriophorum angustifolium</i>, <i>Equisetum fluviatile</i>, <i>E. hymale</i>, <i>E. palustre</i> and <i>Hydrocotyle vulgaris</i> are abundant, whilst <i>Caltha palustris</i> and <i>Cardamine pratensis</i> are less evenly distributed and usually present as scattered individuals. The commonest grass to occur in this community is <i>Molinia caerulea</i>, particularly in drier stands. Bryophytes are almost always conspicuous, especially <i>Scorpidium scorpioides</i> which is a distinctive species of this community.</p>
1a. Ground-water Mire	M10, 7130	<i>Carex dioica</i> , <i>Carex flacca</i> , <i>Carex nigra</i>	<p>Groundwater-fed second-level soligenous mire community of mineral soils and soft, spongy, shallow peats kept very wet by base-rich, calcareous and oligotrophic waters. This is one of the most calcicolous of mire communities in the British Isles and the pH of the flushing waters is mostly high, usually between 5.5 and 7.0 or sometimes higher. It is found in small stands, often associated with spring and rill vegetation within grasslands and more occasionally within ombrogenous</p>

			<p>mires, around topogenous mires and within open water transitions around lakes. It is associated with cool, wet climates. Typically the in situ formation of peat is limited. The community occurs in unenclosed upland environments and most of the stands are grazed and trampled by large herbivores. It is these factors combined with nutrient impoverishment and the strong scouring effect of irrigation, which play a major part in maintaining the community in its generally rich, varied and open state. Most stands would probably progress to scrub or woodland if grazing were withdrawn. The peat substrate is very soft therefore only during periodic dry spells will the vegetation be damaged by trampling and grazing. Where the community runs onto firmer peats around the margins of lakes or basins the effect of grazing may favour the spread of <i>Juncus effusus</i>.</p> <p>This heterogeneous community includes a range of distinctive calcicolous flush vegetation dominated by small sedges including <i>Carex dioica</i>, <i>C. flacca</i> and <i>C. nigra</i> with scattered poor-fen herbs over a patchy carpet of moderately base-tolerant <i>Sphagnum</i> spp. Some grasses occur frequently such as <i>Anthoxanthum uva-ursi</i> and <i>Juncus effusus</i> which can be frequent. Other herbs generally occur as scattered plants, but the most frequent are <i>Equisetum palustre</i>, <i>Hydrocotyle vulgaris</i>, <i>Lotus corniculatus</i>, <i>Potentilla tabernaemontani</i>, <i>Ranunculus flammula</i>, <i>Succisa pratensis</i> and <i>Trifolium pratensis</i>. Other species are characteristic of particular sub-communities. <i>Sphagnum</i> sp. are always obvious, often comprising over half the ground cover.</p>
1b. Sedge Mire	M13, 6410	<p><i>Molinia caerulea</i>, <i>Phragmites australis</i>, <i>Schoenus nigricans</i></p>	<p>Lowland second-level mire community confined to peat or mineral soils irrigated by moist, base-rich, highly calcareous, and oligotrophic to slightly mesotrophic waters. It is often found below springs and seepage lines. This community marks out soils which are kept moist for most of the year and have a moderate to high base-status and the flushing waters are typically pH 6.5 to 8. The structure and floristics of this community are often influenced by grazing and some stands have been affected by a unique and complex history of grazing, mowing and burning. The most prominent structural element, typically rushes and sedges of moderate stature, appear naturally as a rank sward can be kept severely in check by grazing. Shallow peat-digging has been locally important in providing a suitable habitat for the community but more drastic treatment of mires, particularly draining and eutrophication, have reduced its extent and eliminated it from some areas.</p> <p><i>Schoenus nigricans</i> is very frequent and is consistently associated with other common floristic features including <i>Poa flexuosa</i> and <i>Salix repens</i>. Of the sedges, the most striking is <i>Carex acutiformis</i>, which can be frequent or occasionally dominant. The sedge <i>Molinia caerulea</i> is also a constant dominant and, together with the common species, forms a fairly high rough sward with smaller herbs growing in between. In summer this grass, rush and sedge layer can be overtopped by flowering dicotyledons, the most frequent being <i>Filipendula ulmaria</i>. Where the summer water table is close to the surface, species such as <i>Mentha aquatica</i> occur, and <i>Phragmites australis</i> in ungrazed stands. A variety of orchids are found, particularly <i>Dactylorhiza incarnata</i>. On drier areas and particularly tops of <i>Schoenus nigricans</i> tussocks less calcicolous plants are found, most frequently <i>Potentilla erecta</i>.</p>
2a. Rush Pasture	M28, 7230	<p><i>Hydrocotyle vulgaris</i>, <i>Lolium perenne</i>, <i>Stellaria alsine</i></p>	<p>Lowland second-level rush-pasture community characteristic of moist, slightly acidic to neutral, peaty and mineral soils in cool and wet climes. This oceanic community of gently-sloping ground can be found in wetter hollows in the freshwater seepage zone on raised beach platforms and along the upper edge of saltmarshes in sheltered sea lochs, around the margins of soligenous flushes and topogenous mires, as well as on ill-drained, comparatively unimproved or reverted pasture. It can be found on a variety of slightly acidic to neutral soils with a pH of 4 to 7 that are kept moist to wet for most of the year. This community is fairly stable and is maintained mainly by light grazing and more</p>

			<p>occasionally by mowing, which prevents woodland succession. Draining, fertilising and reseeding, however, have reduced its former extent in some areas.</p> <p>This vegetation is ill-defined and characterised by rushes amongst mesophytic herbs common in moist agricultural grassland. The rushes, including <i>Eleocharis palustris</i>, often have a high cover but they may also be sparse. The grass <i>Lolium perenne</i> is frequently important whilst <i>Anthoxanthum odoratum</i> is common in drier stands. Sedges, however, are infrequent with only <i>Carex nigra</i> in existence. There is a variety of frequently occurring common herbs that form mats with high local cover including <i>Hydrocotyle vulgaris</i>, <i>Mentha aquatica</i>, <i>Ranunculus flammula</i> and <i>Stellaria alsine</i>. On patches of wet and open ground, annuals may be prolific, and on cattle-poached mud, <i>Ranunculus sceleratus</i>. Bryophytes are variable in their cover and where the vegetation is open they may be abundant.</p>
2b. Rush Mire	M23, 3130	<i>Carex demissa</i> , <i>Eriophorum angustifolium</i> , <i>Schoenus nigricans</i>	<p>Lowland second-level mire community confined to peat or mineral soils strongly irrigated by base-rich, highly calcareous, and oligotrophic waters ranging from pH 6.5 to 8. This rush-dominated community is located along groundwater seepage lines, in open water transitions around lakes and beside springs and flushes. Grazing primarily influences the structure of this community, while mowing and burning are also important. In contrast, shallow peat digging has been locally significant in providing suitable habitat for this community. Although edaphic characteristics and cool, wet climatic conditions exert a strong influence on the structure and composition of the vegetation, heavy grazing plays a major role in maintaining the distinctive richness of the community, whilst trampling and cropping by livestock are responsible for its unique floristic nature.</p> <p>In this community <i>Schoenus nigricans</i> dominates, giving a grey-green appearance to the vegetation, and is only associated with a small number of other distinctive floral species. Commonly it is intermixed with <i>Eriophorum angustifolium</i> and sedges such as <i>Carex acutiformis</i> and <i>C. demissa</i> which are frequently important. Where groundwater reaches the surface in the summer, water tolerant species such as <i>Equisetum fluviatile</i> occur. Bryophytes species vary in cover in open patches amongst the <i>Schoenus</i> stands, but can be very extensive.</p>

Appendix B. Lake plant communities identified during the June 2012 and 2013 surveys assessed through TWINSPAN, their associated species, National Vegetation Classification (NVC) and Natura 2000 'EUR27' codes (Elkington et al, 2001; EC, 2007).

Community	NVC, EU27	Indicator Species	Description
1. Littoral Coastal	E, 3130	<i>Juncus bulbosus</i> , <i>Nitella translucens</i> , <i>Potamogeton gramineus</i>	Lowland (0-100m.a.s.l.), coastal, circumneutral littoral lake community comprising a high diversity of submerged and floating-leaved macrophyte species indicative of moderately deep areas, including <i>Juncus bulbosus</i> , <i>Nitella translucens</i> and <i>Potamogeton gramineus</i> . Other common species include <i>Isoetes lacustris</i> , <i>Myriophyllum alterniflorum</i> , <i>Potamogeton lucens</i> and <i>Sparganium angustifolium</i> . The pH is circumneutral and above, with moderate conductivity and alkalinity. This community is indicative of oligotrophic to slightly mesotrophic standing waters adjacent to calcareous terrestrial habitats including machair that are often fed by calcareous groundwater. Rare species include the nationally rare <i>Najas flexilis</i> .
2. Benthic Coastal	E, 3140	<i>Chara aspera</i> , <i>Littorella uniflora</i>	Lowland (0-100m.a.s.l.), coastal, circumneutral, benthic lake community comprising a high diversity of stunted macrophyte species indicative of shallow areas, including <i>Littorella uniflora</i> and <i>Chara aspera</i> . The pH is circumneutral and above, with moderate conductivity and alkalinity. This community is indicative of oligotrophic to slightly mesotrophic standing waters adjacent to calcareous terrestrial habitats including machair that are often fed by calcareous groundwater.
1a. Benthic Lowland	C2, 3110	<i>Isoetes lacustris</i> , <i>Nitella opaca</i> , <i>Scorpidium scorpioides</i>	Lowland to slightly upland (<250m.a.s.l.), slightly acid, benthic lake community comprising a high diversity of stunted macrophyte species indicative of shallow areas including <i>Isoetes lacustris</i> , <i>Nitella opaca</i> and <i>Scorpidium scorpioides</i> . <i>Juncus bulbosus</i> is constant, often with <i>Chara aspera</i> and <i>Nitella translucens</i> . This community is indicative of oligotrophic to slightly mesotrophic standing waters with low conductivity and alkalinity, although may be found in areas where sandy minerals are present including near terrestrial machair habitats.
1b. Littoral Lowland	I, 1130	<i>Myriophyllum spicatum</i> , <i>Potamogeton crispus</i>	Lowland (0-75m.a.s.l.), base-rich, littoral lake macrophyte community of coastal locations dominated by <i>Myriophyllum alterniflorum</i> and <i>Potamogeton crispus</i> . This moderately species-rich assemblage also includes <i>Potamogeton gramineus</i> and <i>Sparganium angustifolium</i> indicative of oligotrophic to slightly mesotrophic calcareous conditions due to surrounding terrestrial habitats such as sand dunes and machair. The pH is relatively high, with moderate conductivity and high alkalinity.
2a. Shallow Coastal	E, I, 3140	<i>Chara aspera</i>	Lowland, coastal, circumneutral benthic lake macrophyte community indicative of shallow areas dominated by <i>Chara aspera</i> . Lake waters have a circumneutral pH, with moderate conductivity and alkalinity. The community indicates oligotrophic to mesotrophic conditions and is often found adjacent to calcareous terrestrial habitats such as machair.
2b. Shallow Lowland	B, 3140	<i>Littorella uniflora</i> , <i>Nitella opaca</i> , <i>Potamogeton natans</i>	Lowland (<200m.a.s.l.), slightly acidic, moorland, macrophyte community indicative of small shallow lakes, with a limited range of plants including <i>Littorella uniflora</i> , <i>Nitella opaca</i> and <i>Potamogeton natans</i> . This community is found in standing oligotrophic waters adjacent to peatland that have a low conductivity and alkalinity, and also contain very few minerals of sandy plains.

Aquatec Equipment Awards 2012 Report

Hydrogeomorphology and ecology in a sedimentary coastal lake and wetland system: Sheskinmore Lough SPA

Introduction

Sheskinmore Lough Special Protection Area (SPA), Donegal, northwest Ireland (Figure 1), comprises a shallow sedimentary lake surrounded by a diverse array of coastal habitats supporting a plethora of endangered species, including the protected Slender Naiad (*Najas flexilis*). Since its formation (c.1000yrBP), the lake and its surroundings have undergone significant changes contributing to the present complex hydrogeology and associated ecosystem (NPWS 2005). In particular, the eco-hydrological regime of the lake and wetland is complicated by the sedimentary drape of calcareous dune sand over metamorphosed granitic bedrock (NPWS 2005). Habitats listed in the site designation include sedimentary shallow lake, calcareous grassland, saltmarsh, intertidal sand flats, swamp, fen and wet grassland, which all support a plethora of rare and endangered plant and bird species many of which are protected under EU legislation. The lough itself formed c. 1000 years ago when dune remobilisation separated the Duvoge and Abberachrin river branches from the main Loughros More estuary, resulting in impoundment of these small rivers and development of a shallow lake (Shaw & Carter 1994).

Since its isolation, salinity has decreased within the lake and, although it is now classed as freshwater, the underlying acidic bedrock and seepage of basic water from the calcareous dunes presents a challenging and complex hydro-geological regime and associated ecosystem structure. The National Parks and Wildlife Service (NPWS) have suggested that water levels have declined significantly in recent years, citing natural siltation and ad hoc drainage works as causes, and negative impacts on roosting habitat for protected bird species as an unwelcome consequence (Emer Magee pers. comm. 11th February 2012). Management of these effects has been through the installation of a sluice to maintain water levels, but the paucity of knowledge and understanding of lake evolution, current ecology and related hydro-geomorphic controls, in addition to the natural ecological variability of the system, suggests that continued reactionary management is unlikely to result in successful and sustainable management of this internationally important system.



Figure 1. Sheskinmore Lough SPA and the position of the lake and rivers relative to the Loughros More estuary (OSI 2011).

The interdisciplinary PhD that this report is linked to will use an array of techniques (palaeolimnological, ecological sampling, bathymetric/topographic surveys, hydrological modelling) to investigate the influence of these changes on the site ecology. By examining these impacts in a climate change and conservation context, the PhD hopes to improve our understanding of coastal lakes and enhance their management on an international scale. This report provides a valuable starting point for these analyses and aims to develop a more detailed understanding of the hydrology of Sheskinmore Lough by investigating lake level fluctuations in relation to precipitation and temperature. The findings will not only enhance understanding of the lake hydrology, they will also contribute to the wider PhD study and ultimately increase knowledge regarding the entire site.

Methods

The study was conducted over a six-week period from 6th September 2012 to 23rd October 2012. Hydrological data were collected using a network of automatic loggers that measure water and atmospheric pressure (Figure 2). A Rugged BaroTROLL logger was used to measure atmospheric pressure at the site, 3 Rugged TROLL 100 automatic loggers were deployed to measure water depth (pressure head above sensor) at 15-minute intervals in the two inflows (Duvoe and Abberachrin rivers) and outflow (at the sluice), and an Aquatec AQUAlogger 520PT10 temperature and depth logger was deployed in the south eastern side of the lake in order to measure water level and temperature variations at 15-minute intervals (Figure 3). The TROLLS in the inflows were positioned inside protective plastic tubing and secured by concrete blocks to ensure they remained upright and stationary in flowing water. The AQUAlogger and the TROLL at the sluice were

also positioned inside protective tubing, but were secured to existing structures (submerged fence post and sluice railings, respectively).



Figure 2. Position of the weather station and the four monitoring sites in the inflows, lake and outflow at Sheskinmore Lough SPA (OSI 2011).



Figure 3. Photograph showing the AQUAlogger 520PT10 deployed in the south eastern side of Sheskinmore Lough on 6th September 2012.

The four water pressure loggers were programmed to accommodate freshwater density prior to deployment. Data were processed using standard pressure-depth conversion calculations. The TROLL and AQUAlogger pressure data were converted to water depth (h, measured in metres) using:

$$h = \frac{P_o - P_a}{100g\rho_w}$$

where P_o is logged pressure (mbar), P_a is atmospheric pressure (mbar), g is acceleration due to gravity (9.81 m/s²) and ρ_w is (fresh) water density (1,000 kg/m³)

The depth calculations were then converted to water level (distance between water and ground surface) using manual measurements obtained during instrument deployment. On completion of these calculations, it was noted that the AQUAlogger operated with a higher than normal sensitivity to temperature. Hydrostatic pressure ($P_o - P_a$) was therefore pre-processed to remove this cyclic water temperature influence. Raw atmospheric pressure and temperature data readings were used to calculate a linear correction factor for the pressure variation with temperature. The raw readings were then normalised and calibration coefficients applied to correct the hydrostatic pressure data and the above conversions applied. Precipitation and air temperature data were recorded at 30-minute intervals using a Davis Weatherlink VantagePro2 meteorological station positioned approximately 750m to the southeast of the lake. Precipitation data were converted to 15-minute intervals by interpolating between readings to allow for direct comparison with depth data. The sluice, which is manually operated by the NPWS, was left open for the duration of the study, allowing free movement of water from the lake to the estuary.

Results and Discussion

The results of this study show that the lake level of Sheskinmore Lough varies in direct accordance with precipitation (Figure 4). Over the six-week period the AQUAlogger 520PT10 recorded the transition from lower summer to higher autumn water levels (0.30m to 0.60m), which were accentuated by a dramatic increase in levels (~0.40m) within 24hrs from around 0.30m to around 0.70m following a high intensity rainfall event of 5.2cm on 12th September 2012 (Figure 5). This seasonal shift in water level from low to higher levels is likely linked to declines in temperature and, consequently, lower evaporation rates at the end of September (Lenters 2001; Zohary & Ostrovsky 2011). Several sporadic and low intensity (<2cm) precipitation events from 13th to 29th September 2012 maintained the lake water above 0.65m. In contrast, a series of higher intensity (>2cm), high frequency (daily) rainfall events occurring in close succession between 30th September and 5th October 2012 resulted in another fairly marked increase in lake level to 0.95m (Figure 6). A dry spell from 6th to 10th October 2012 and a series of fairly low intensity (<2cm) and low frequency (every other day) precipitation events for the remainder of the study period, however, meant that lake levels gradually dropped to levels of 0.60m.

Water level fluctuations in the inflows and outflow also exhibit strong relationships with precipitation events, however the data reveal contrasting responses. For example, the sluice outflow TROLL, although positioned at a greater depth than the AQUAlogger, displays an identical time series to the lake data, with

the exception of its response to a low intensity but high frequency period of precipitation from the 10th to 11th September 2012 (Figure 7). This intense rainfall period resulted in a rapid increase in water levels at the sluice from around 0.90m to 1.60m in only 24 hours. This is in contrast to an increase of 13cm across the lake over the same time period. Similarly, the Duvoe TROLL also reveals comparable levels to the lake. Its relatively close proximity (0.5km) means there is likely to be an 'upstream effect' as increasing water depth and expanding lake area forces water levels to rise upstream (Baxter 1977; Parker et al 2004). The presence of a wetland and therefore saturated substrate in this area enhances this effect and riverine stream flow water levels are consequently disguised.

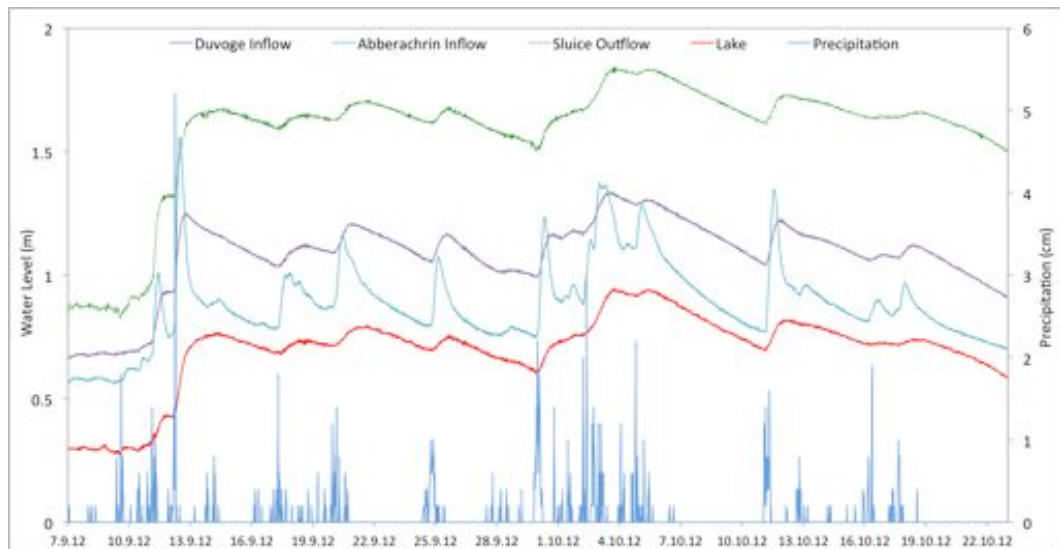


Figure 4. Variation in precipitation and water level at the four Sheskinmore Lough monitoring sites over 15-minute time intervals between 00:00hrs on 7th September 2012 and 23:45hrs on 22nd October 2012.

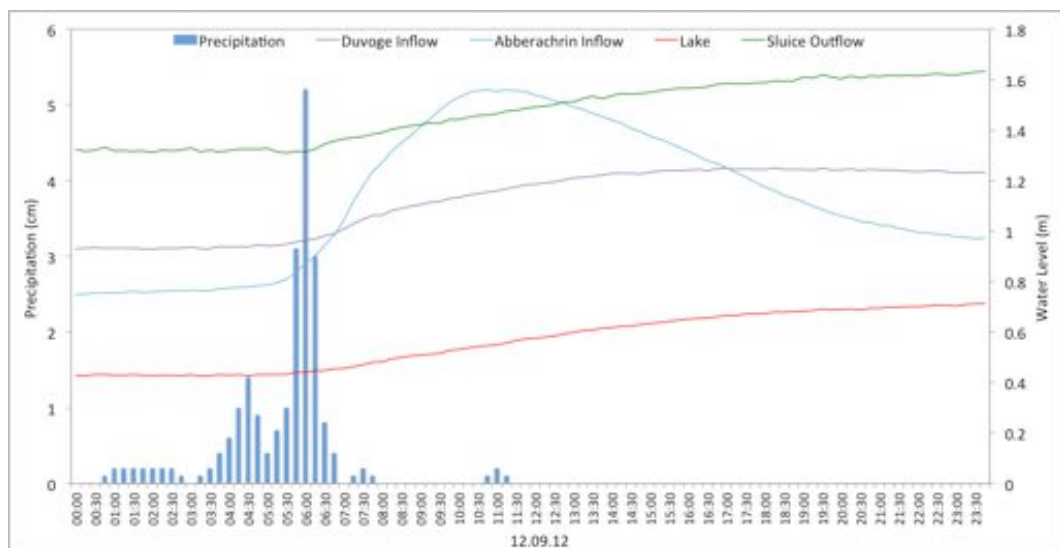


Figure 5. Variation in precipitation and water level at the four Sheskinmore Lough monitoring sites over 15-minute time intervals between 00:00hrs and 23:45hrs on 12th September 2012.

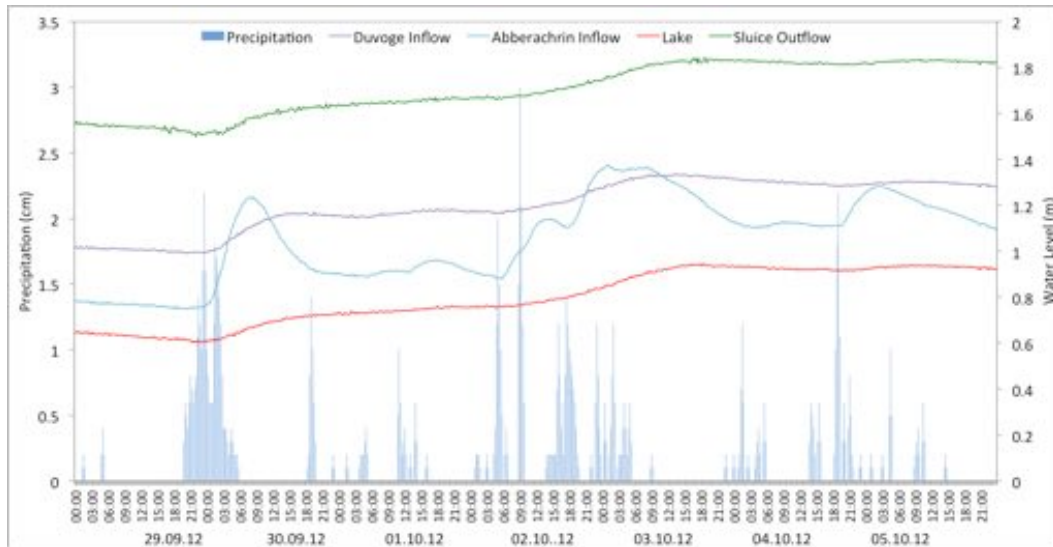


Figure 6. Variation in precipitation and water level at the four Sheskinmore Lough monitoring sites over 15-minute time intervals between 00:00hrs on 29th September 2012 and 23:45hrs on 5th October 2012.

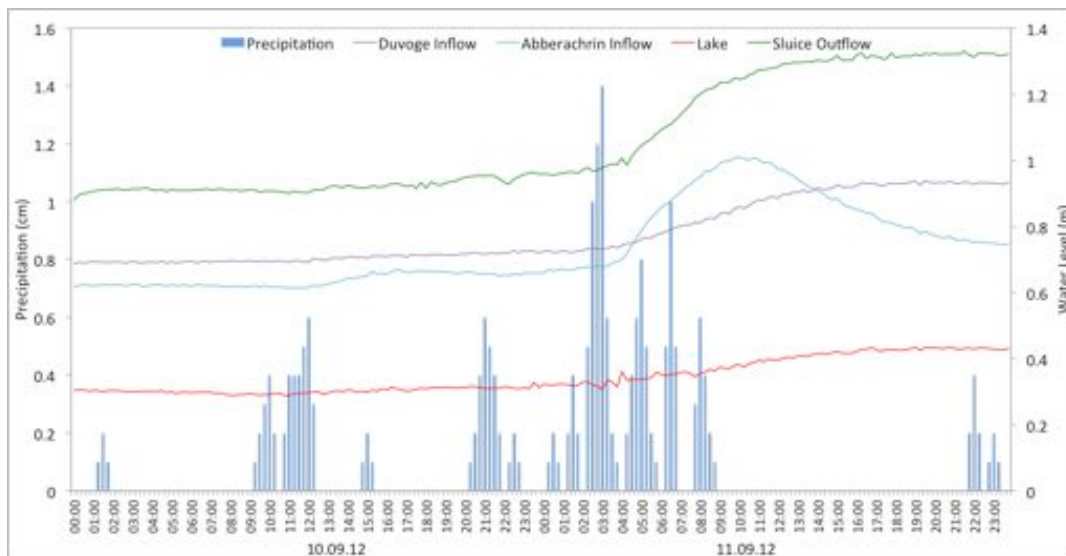


Figure 7. Variation in precipitation and water level at the four Sheskinmore Lough monitoring sites over 15-minute time intervals between 00:00hrs on 10th September 2012 and 23:45hrs on 11th September 2012.

The Abberachrin and Duvoe inflow time series display very different water level fluctuations. As previously identified, the water level variation in the Duvoe is very similar to that recorded by the AQUAlogger. The relationship between the Duvoe and lake water levels is strong, showing a positive correlation ($r^2 = 0.928$), as opposed to the Abberachrin, which shows little relationship ($r^2 = 0.478$). Again this is likely a result of the proximity of inflow monitoring stations to the lake itself (Baxter 1977; Parker et al 2004). The Abberachrin TROLL, located 1.5km from the lake, therefore has revealed a time series typical of a true riverine response to precipitation. Although the Duvoe inflow exhibits water level data that, overall, resembles the pattern in lake level variation, two noticeable differences are apparent that indicate its riverine location. Firstly, greater overall response of the Duvoe water level to low intensity rainfall events in comparison to the lake water level when water depth fluctuations are considered across 24-hour periods (Figure 7). Reduced channel area upstream of the lake means the same amount of

precipitation falling at the Duvoge monitoring site will result in greater volumes of water where accommodation space is extremely limited (Euliss & Mushet 1996; Coops et al 2003). Secondly, water levels in the Duvoge respond more readily to high intensity and low intensity-high frequency precipitation events than the lake itself. For example, the high intensity (>2cm) rainfall event on 30th September resulted in a 0.10m increase in water level in the Duvoge in only 12 hours, compared to a 0.10m increase over 18 hours in the lake (Figure 6). Again this is a function of accommodation area, although when larger volumes of precipitation over longer timescales are concerned, water level is determined, not just by channel cross section, but also by catchment area and substrate saturation (Gibson et al 2006; Coops & Hosper 2009).

Nevertheless, it should be noted that the Duvoge water level variation is very similar to the lake level, suggesting it has an important influence on the water level regime of Sheskinmore Lough. The Abberachrin, on the other hand exhibits a far more 'flashy' water level regime typical of riverine response to precipitation (Figure 4) (O'Sullivan & Reynolds 2003; Alemayehu et al 2006). This is likely due to the Duvoge flowing within a wider peaty valley where as the Abberachrin resides within a narrow bedrock valley. Consequently, water levels in the Abberachrin increase rapidly following precipitation events, predominantly when rainfall exceeds 1cm for a period of over an hour. Water levels in this inflow river also decline more rapidly than the Duvoge, lake and outflow during dry periods following rainfall events. As a general rule it can be said that the higher the rainfall, the flashier the response of the Abberachrin, with increased levels of 0.5-1m following intense rainfall events (>1.5cm) as opposed to 0.20m increases following minimal rainfall events (<1.5cm).

An exploration of the daily average precipitation and water level (Figure 8) reveals that, although 12th September 2012 experiences the highest rainfall period, 2nd October 2012 exhibits the greatest average precipitation due to sustained precipitation for 6 hours peaking at 3.6cm. The number of rain days exceeds the number of rain-free days by 33 to 13, with 14 days experiencing average rainfall of over 1cm. Table 1 reveals that precipitation is highly variable over short timescales, with a range of 5.2cm and a mean of 0.7cm when the 15-minute interval data are considered, as opposed to a range of 3.6cm and a mean of 0.3cm when rainfall is averaged over each day. It should be noted that the Duvoge inflow shows greater variation in water level in the lower resolution data in Figure 8. The enhanced fluctuations in this time series are not only more akin to the flashier Abberachrin river time series, they also emphasise the extremes in water level occurring in the Duvoge that are less evident in the higher resolution data, highlighting its variable riverine regime.

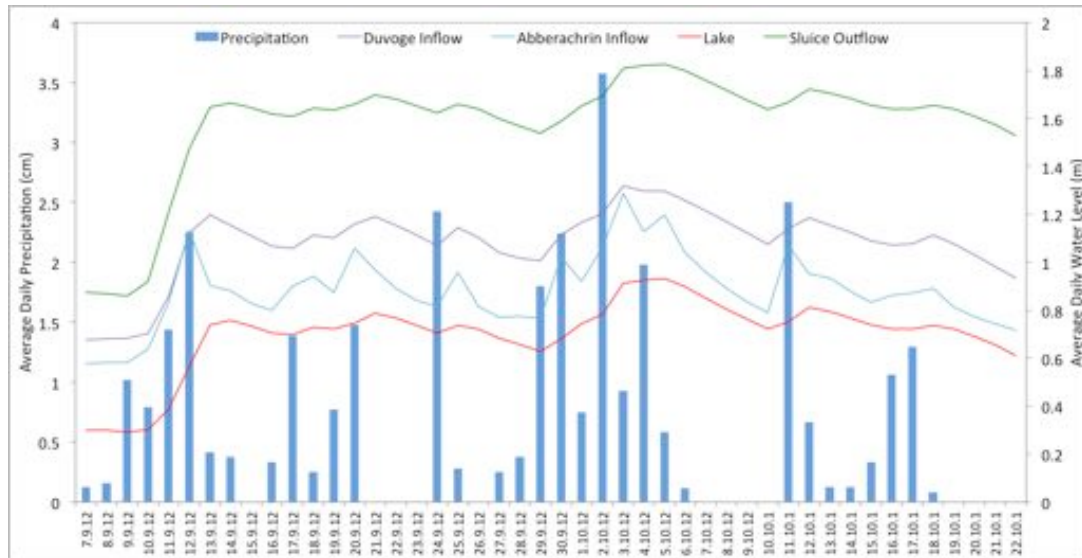


Figure 8. Average daily variation in precipitation and water level at the four Sheskinmore Lough monitoring sites from 7th September 2012 to 22nd October 2012.

The range of water level variation in the four monitoring sites at Sheskinmore Lough is displayed in Figure 9 and Table 2. These analyses reveal the Abberachrin inflow as the most variable of the sites in terms of quartile range for both high resolution (15-minute) and low resolution (daily) timescales; however when the entire water level range is considered, the outflow has experienced the greatest fluctuation. The Duvoge inflow exhibits the same range as the lake for both the high (0.68m) and low resolution (0.64m) timescales; however the quarterly range reveals the Duvoge water level varies slightly more than the lake level.

Table 1. Summary of precipitation data (15-minute intervals) and average daily precipitation data at Sheskinmore Lough over the 7th September 2012 to 22nd October 2012 study period.

	Precipitation (cm)	Average Daily Precipitation (cm)
Max	5.2	3.6
Upper Quartile	1.8	1.1
Mean	0.7	0.3
Lower Quartile	0	0
Min	0	0
Quartile Range	1.8	1.1
Range	5.2	3.6

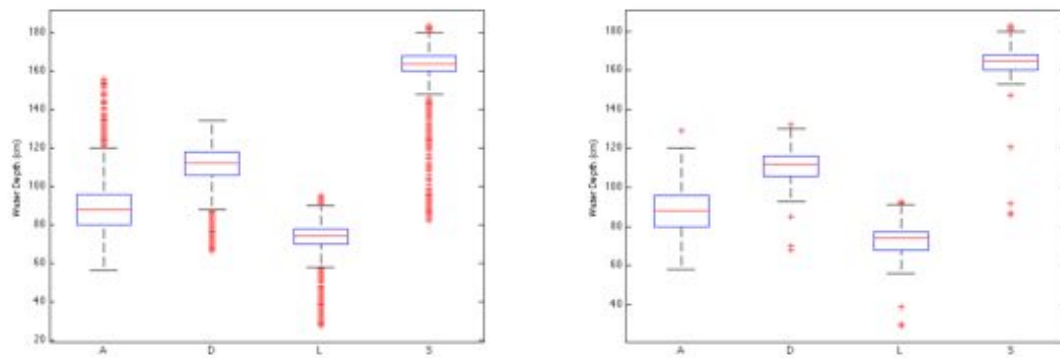


Figure 9. Boxplots showing the range of water level (15-minute interval) and range of average daily water level data at the four monitoring sites around their respective instrument depths at Sheskinmore Lough over the 7th September 2012 to 22nd October 2012 study period (A: Abberachrin inflow, D: Duvoige inflow, L: lake, S: Sluice outflow).

At first it appears there is a fairly strong positive relationship between lake and air temperature as both exhibit the same overall declining trend during the study period and both have a similar average temperature of 12.8°C and 12.5°C, respectively (Figure 10); however when the data is explored in greater detail it reveals a weak relationship ($r^2 = 0.324$). This is likely in part due to low wind speeds and therefore minimal mixing within the water column, but is predominantly the result of the slower response time of water to heat conductivity than air and therefore the lake takes longer to heat up than the surrounding atmosphere (Dingman & Bedford 1984; Wuest & Carmack 2000). Overall, air temperature and the lake water temperature both rise and fall in a daily cycle of day and night, respectively.

Table 2. Summary of water level (15-minute intervals) and daily average water level data at the four monitoring sites at Sheskinmore Lough over the 7th September 2012 to 22nd October 2012 study period.

Water Level (m)	Abberachrin Inflow	Duvoige Inflow	Lake	Sluice Outflow	Average Daily Water Level (m)	Abberachrin Inflow	Duvoige Inflow	Lake	Sluice Outflow
Max	1.56	1.34	0.95	1.84	Max	1.29	1.32	0.93	1.83
Upper Quartile	0.95	1.17	0.77	1.68	Upper Quartile	0.96	1.16	0.77	1.68
Mean	0.87	1.11	0.73	1.64	Mean	0.88	1.11	1.64	0.73
Lower Quartile	0.79	1.05	0.69	1.60	Lower Quartile	0.80	1.06	0.69	1.60
Min	0.56	0.66	0.27	0.82	Min	0.58	0.68	0.29	0.86
Quartile Range	0.16	0.12	0.08	0.08	Quartile Range	0.16	0.10	0.08	0.08
Range	1.00	0.68	0.68	1.02	Range	0.71	0.64	0.64	0.97

Air temperature rises and falls more rapidly than the lake water temperature, exhibiting a greater overall range (18.4°C as opposed to 12.1°C) and quartile range (1.80°C to 0.90°C) (Table 3). This is clearly a function of slow heat conductivity

between air and water: however it does not totally explain why the lake water temperature reaches a greater maximum on 8th September 2012, reaching 20.5°C at 17:30, despite the outside air temperature having dropped from 19 °C to 16 °C. The lake retains its heat for longer due to the greater specific heat capacity of water, despite rapidly decreasing air temperatures (Wuest & Carmack 2000). The lowest temperatures of both air and water occur during early to mid October. The majority of September experiences moderate temperature variation except for during the nights of 21st and 22nd September 2012 when air temperatures dropped to lows of 3°C and 1°C, respectively. Although there is no strong relationship between temperature and lake level, the water depth time series in Figure 10 reveals a loose association. The majority of the highest daily air temperature maxima coincide with dips in water level, therefore reflecting the cumulative influence of evaporation over time from the lake surface. As with the weak relationships between the levels in the inflows, outflows and lake described previously, further investigations into additional hydrological processes such as evaporation are required in order to truly separate the individual roles of each part of the system.

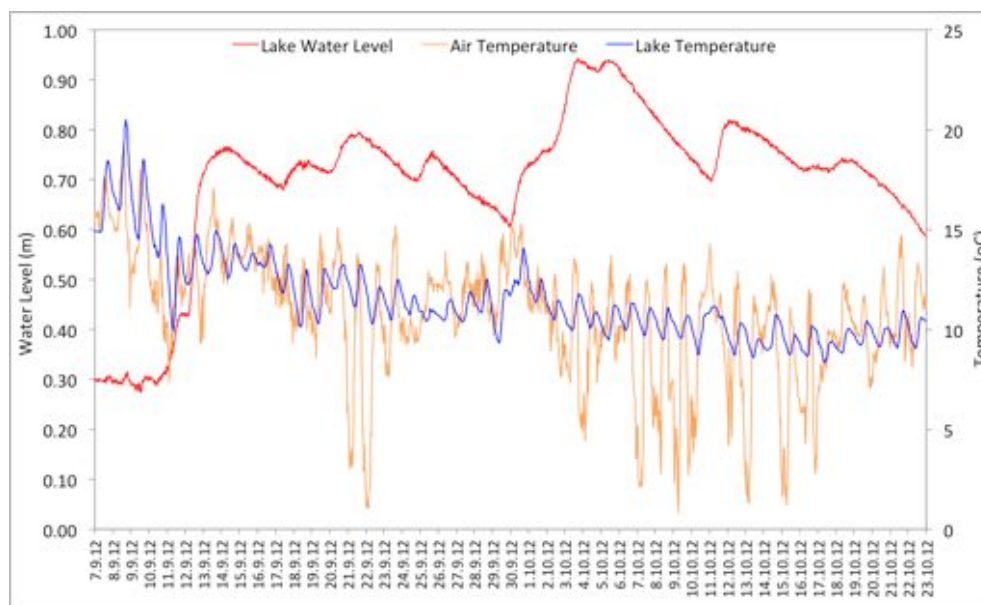


Figure 10. Lake temperature, air temperature and lake water level recorded at 30minute intervals over the 7th September 2012 to 22nd October 2012 study period.

Table 3. Summary of lake water temperature, air temperature and water level (30-minute intervals) data at Sheskinmore Lough over the 7th September 2012 to 22nd October 2012 study period.

	Air Temperature (°C)	Lake Water Temperature (°C)	Lake Water Depth (m)
Max	19.2	20.5	0.94
Upper Quartile	11.2	11.0	0.73
Median	12.8	12.5	0.77
Lower Quartile	9.40	10.1	0.69
Min	0.80	8.40	0.27
Quartile Range	1.80	0.90	0.04
Range	18.4	12.1	0.67

Conclusion

The hydrological system at Sheskinmore Lough consists of a small shallow lake fed by two separate, small riverine systems that exhibit high sensitivity to precipitation. The rapid responses of these rivers and their saturated catchments ensure the water levels in the lake fluctuate considerably and maintain elevated levels even during dry periods. The disconnection between air temperature and lake water temperature indicate insignificant mixing within the lake water column and the data also suggests that evaporation occurs during periods when precipitation is minimal. Although differences exist in water level between the four monitored parts of the system and between air and lake water temperature, the true functioning of each hydrological component across the site is dampened by factors such as the location of monitoring stations and the lack of investigation into other hydrological processes. In order for this PhD to fully understand the hydrological linkages across the SPA, further research will be required, with analyses spanning the wetland system as far out as the estuary. More in depth analysis of both hydrological and meteorological processes is vital if we are to identify the importance of the various hydrological components and therefore determine the positive and negative impacts of anthropogenic practices at Sheskinmore Lough.

Ultimately this report has provided a valuable starting point for hydrological analyses at Sheskinmore Lough SPA. The AQUAlogger 520PT10 has been invaluable in aiding the development of a more detailed understanding of the hydrology of Sheskinmore Lough. With the valuable addition of temperature data and, when used in conjunction with the existing TROLL loggers, the AQUAlogger has acted as a vital springboard for this PhD in terms of directing further research, including investigations focussed across the entire SPA, not just around the lake itself.

Acknowledgements

I would like to express my sincere thanks to: Aquatec for very generously awarding me the use of the AQUAlogger, especially Alison Callaway and Reva Perryman for liaising with me throughout the whole process, and Andy Smerdon for his help with data conundrums; Andrew Speer, Emer Magee and Tim Roderick of the NPWS for their continual support and for kindly providing a boat to save our feet from getting wet; and last, but not least, my supervisor Dr Helene Burningham for all her constant help and enthusiasm in the field and in the office.

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Appendix D. Full descriptions of the WETMEC hydrological classification groups and subgroups associated with the hydrological monitoring sites across the central and surrounding wetland (W1-7 and S1-5) and dune (D1-2) systems (Wheeler et al, 2009; Kimberley & Coxon, 2013).

Feature	WETMEC 3: Buoyant Weakly Minerotrophic Surfaces (“Transition Bogs”)	WETMEC 6f: Surface Water Percolation Water Fringe	WETMEC 9: Groundwater-Fed Bottoms	WETMEC 12: Fluctuating Seepage Basins	WETMEC 20b: Percolation Water Fringe
Key character combination	Quaking, summer-wet surface elevated slightly above telluric water tables; often in basins, over high or low permeability deposits	Surface usually wet in summer and flooded in winter. Peat top-layer often loose, and encroaching directly up open water	Troughs or basins, usually on quite deep peat; if on floodplains, isolated from river. Marginal springs less evident	Small sumps with strongly fluctuating water table, often from well below surface to flooded, which may relate to aquifer levels	Adjoining open water and receiving water from this, may have different provenance to upslope sources
Landscape	Basins	Floodplains	Floodplains, basins, troughs, valleys	Valley heads and basins	Basins
Topography	Flat	Flat	Flat	Shallow sumps	Flat
Summer water level & main water source	Near or at surface. May receive weakly telluric water, but precipitation probably a significant component of budget	Usually slightly subsurface. Fed mainly by surface water, often from dykes connected to watercourses	Apparently groundwater-fed, but ground water table often well below surface, sometimes because of drainage	Mainly groundwater-fed. Water table variable, depending on topography and aquifer level; fluctuates strongly	Water table at or near surface, fed mainly by surface runoff, some of which sourced by groundwater outflow
Ground water association	Connectivity with aquifers often uncertain. Outflow likely. In some cases may recharge aquifer. Groundwater often sub-surface	Generally unimportant; may sometimes contribute to water level in dykes. Dyke level usually below surface	Aquifer may be episodically at, above or near surface, but is often low (and more or less in equilibrium with wetland water table)	Aquifer episodically at, above or near surface. Water level sometimes in (slow) equilibrium with aquifer level, but relationship sometimes obscure	More or less confined or very minor aquifer, or none; sometimes springs and seepages visible, usually well upslope
Watercourse association	None	Adjoins stands, either as watercourses or watercourse-connected dykes	Often associated with watercourses, but usually isolated from these and (well) above them	Mostly not associated with water courses, but sometimes lateral to water course	Water body irrigates stand. Provenance of water may be different
Upslope surface water association	Some sites have locally significant stream or field-drain inflow in addition to rain generated runoff	Directly adjoins water bodies or connected dykes and may contribute to dyke levels, mainly during winter	May be some rain-generated run-off, but much infiltrates into ground above site, or intercepted by drains	Little evidence for surface water inflows (except where sumps have been connected by drains)	May also receive water from upslope telluric sources
Surface flooding	None	Rare to frequent winter flooding	None	Usually inundated episodically	Normally with surface water
Stand flow	Not visible	Usually not visible	Not visible	Usually none	Sometimes visible
Summer water outflow	Often none	Usually deep, often >4m. Peat, sometimes with thick alluvial intercalations	Sometimes weak outflow visible, or seepage into drains within wetland	Usually none except when water tables are very high	Shallow to deep
Depth of peat & alluvium	Often deep (> 3m), but can be shallow	Spongy, sometimes quaking or semi-floating surface. Top layer of permeable peat over a less permeable layer	Shallow to deep	Very shallow to moderate Amorphous organic material. Variable permeability, but mostly moderate	Shallow to deep
Peat & alluvium permeability	Quaking surface; usually over a similarly quaking peat deposit. Surface peat more permeable than the lower substrata	Most often over low-permeability clays etc. Alluvial deposits sometimes interlayered with peat. Some over sandy deposit	Firm amorphous peat, mostly of moderate permeability	Mostly sands and gravels to sandy clays of moderate permeability; some evidence for low permeability layers in basin lining	Typically very unconsolidated and unstable, but may be rooted swamp rather than buoyant surface
Basal substratum permeability	Variable: from dense clays to gravels. Usually separated by a low-permeability infill		Mostly over sands and sandy clays. Sometimes-local lenses of marl. Fairly permeable		Mostly over clays and silts, or presumed low-permeability bedrock

Appendix E. Exploratory sediment core information including: location, date excavated, number of samples recovered, and detailed cross sectional sedimentology with depth below ground surface.

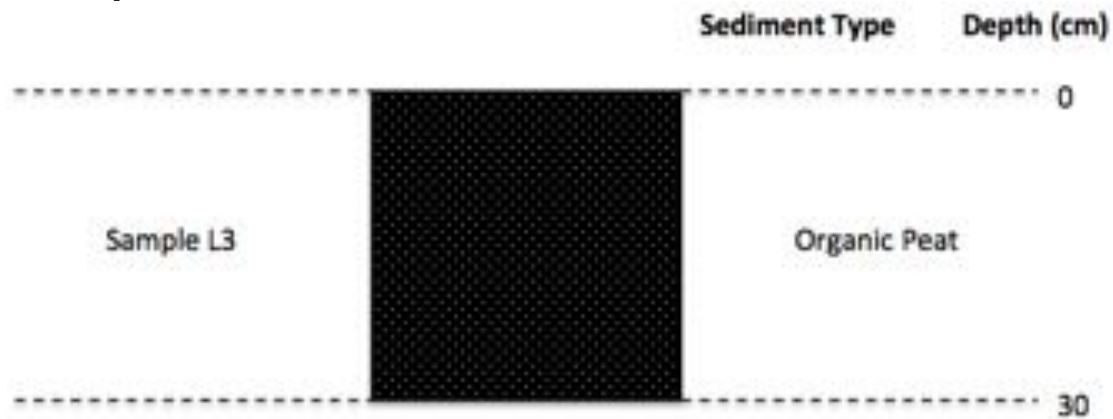
Sheskinmore Sediment Core L3

Coordinates: 54.80908547, -8.46424456

Surface Elevation (mOD): 2.50

Date Excavated: 17th June 2012

No. of Samples Recovered: 1



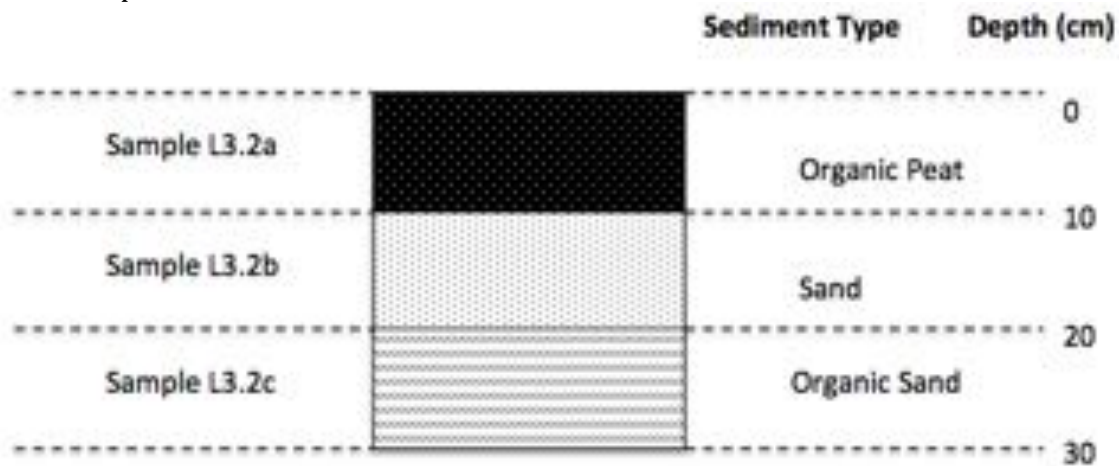
Sheskinmore Sediment Core L3.2

Coordinates: 54.80865967, -8.46635613

Surface Elevation (mOD): 2.70

Date Excavated: 17th June 2012

No. of Samples Recovered: 3



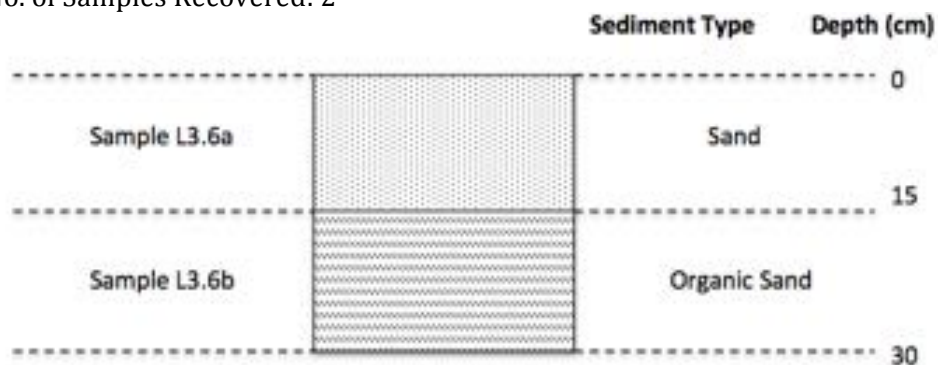
Sheskinmore Sediment Core L3.6

Coordinates: 54.80738763, -8.46782028

Surface Elevation (mOD): 2.80

Date Excavated: 17th June 2012

No. of Samples Recovered: 2

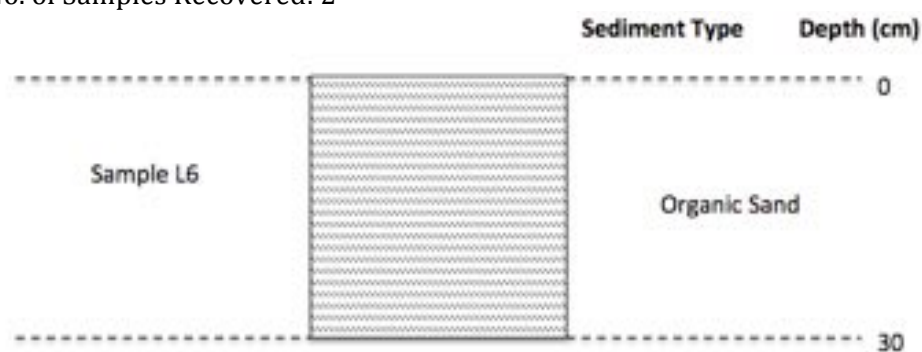
**Sheskinmore Sediment Core L6**

Coordinates: 54.80234021, -8.45323132

Surface Elevation (mOD): 5.81

Date Excavated: 17th June 2012

No. of Samples Recovered: 2

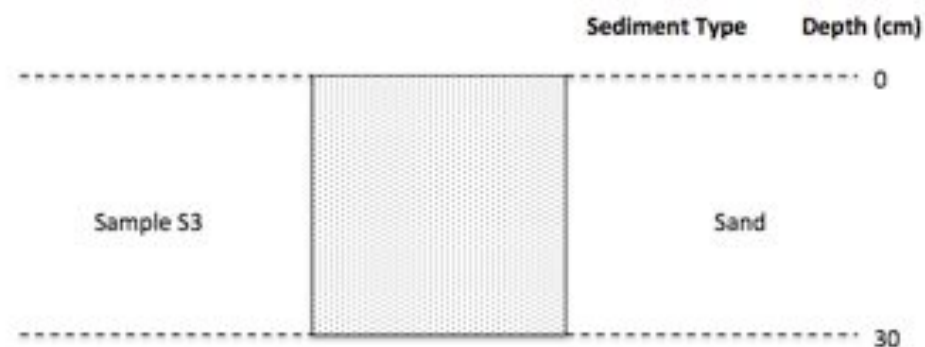
**Sheskinmore Sediment Core S3**

Coordinates: 54.809919, -8.479228

Surface Elevation (mOD): 8.32

Date Excavated: 17th June 2012

No. of Samples Recovered: 1



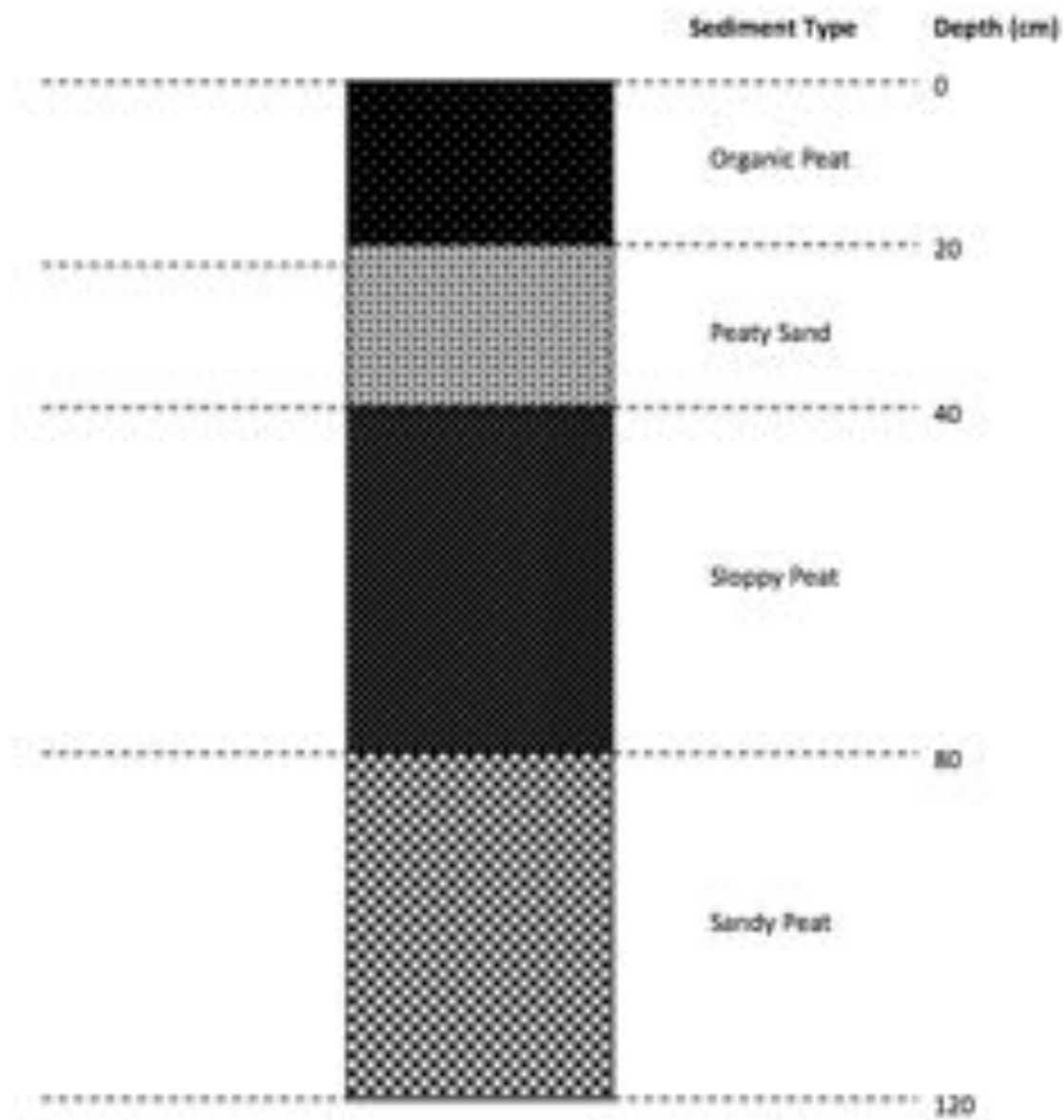
Sheskinmore Sediment Core W1

Coordinates: 54.806703, -8.482431

Surface Elevation (mOD): 6.83

Date Excavated: 17th June 2012

No. of Samples Recovered: 0



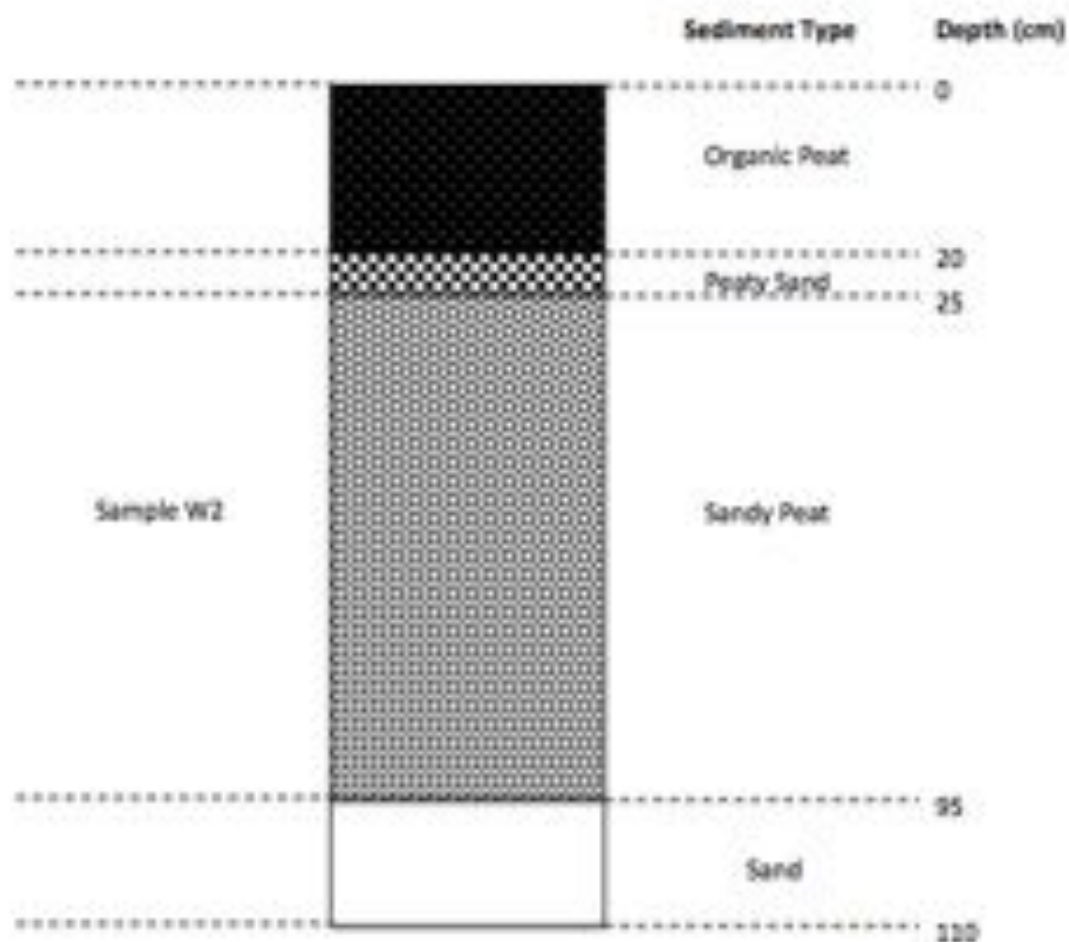
Sheskinmore Sediment Core W2

Coordinates: 54.807751, -8.478482

Surface Elevation (mOD): 5.02

Date Excavated: 17th June 2012

No. of Samples Recovered: 1



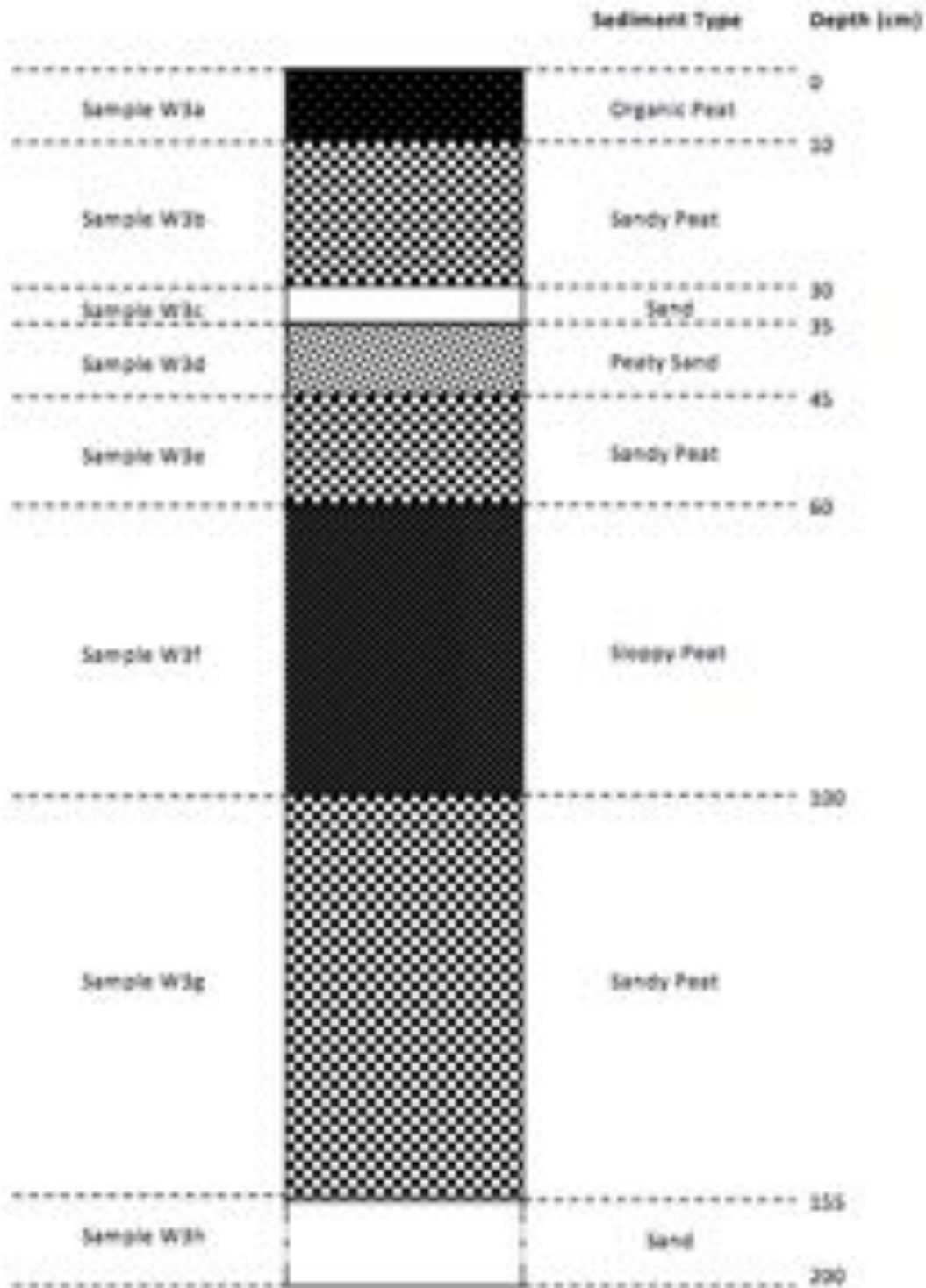
Sheskinmore Sediment Core W3

Coordinates: 54.808481, -8.474679

Surface Elevation (mOD): 3.16

Date Excavated: 17th June 2012

No. of Samples Recovered: 8



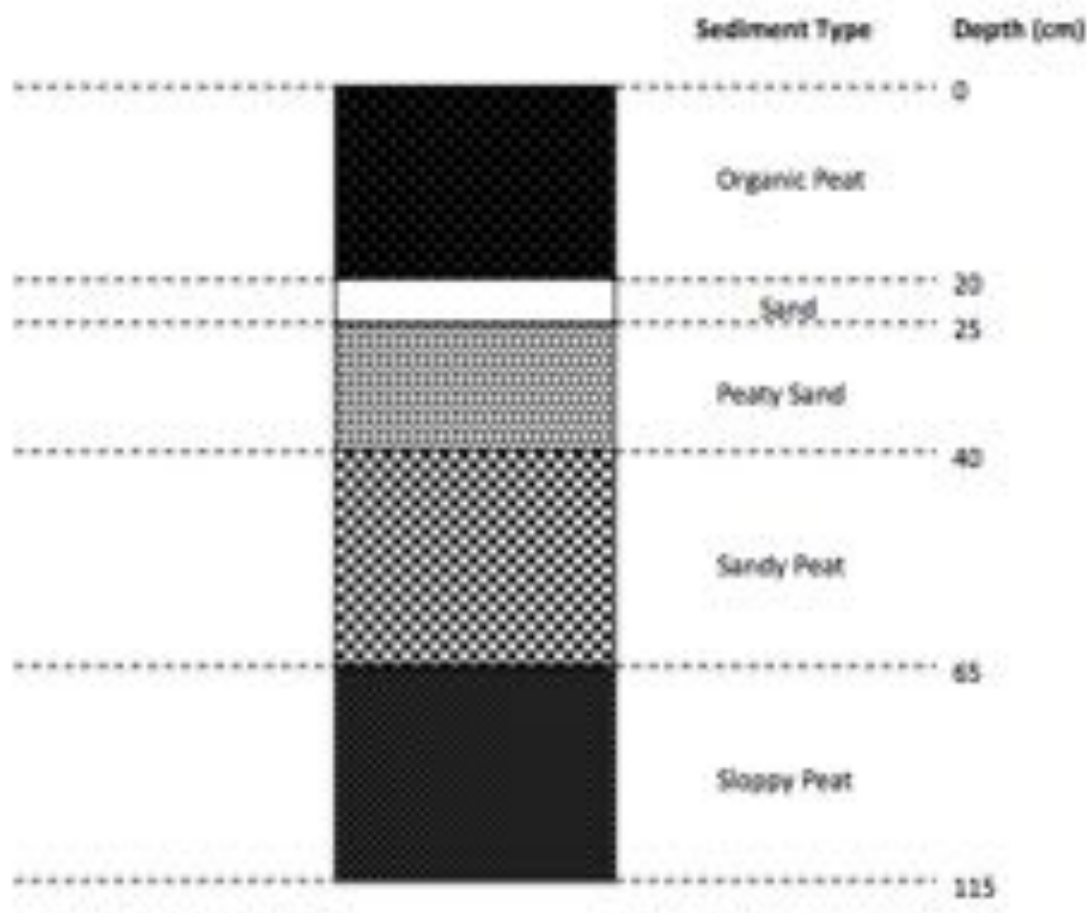
Sheskinmore Sediment Core W4

Coordinates: 54.808646, -8.473666

Surface Elevation (mOD): 3.44

Date Excavated: 17th June 2012

No. of Samples Recovered: 0



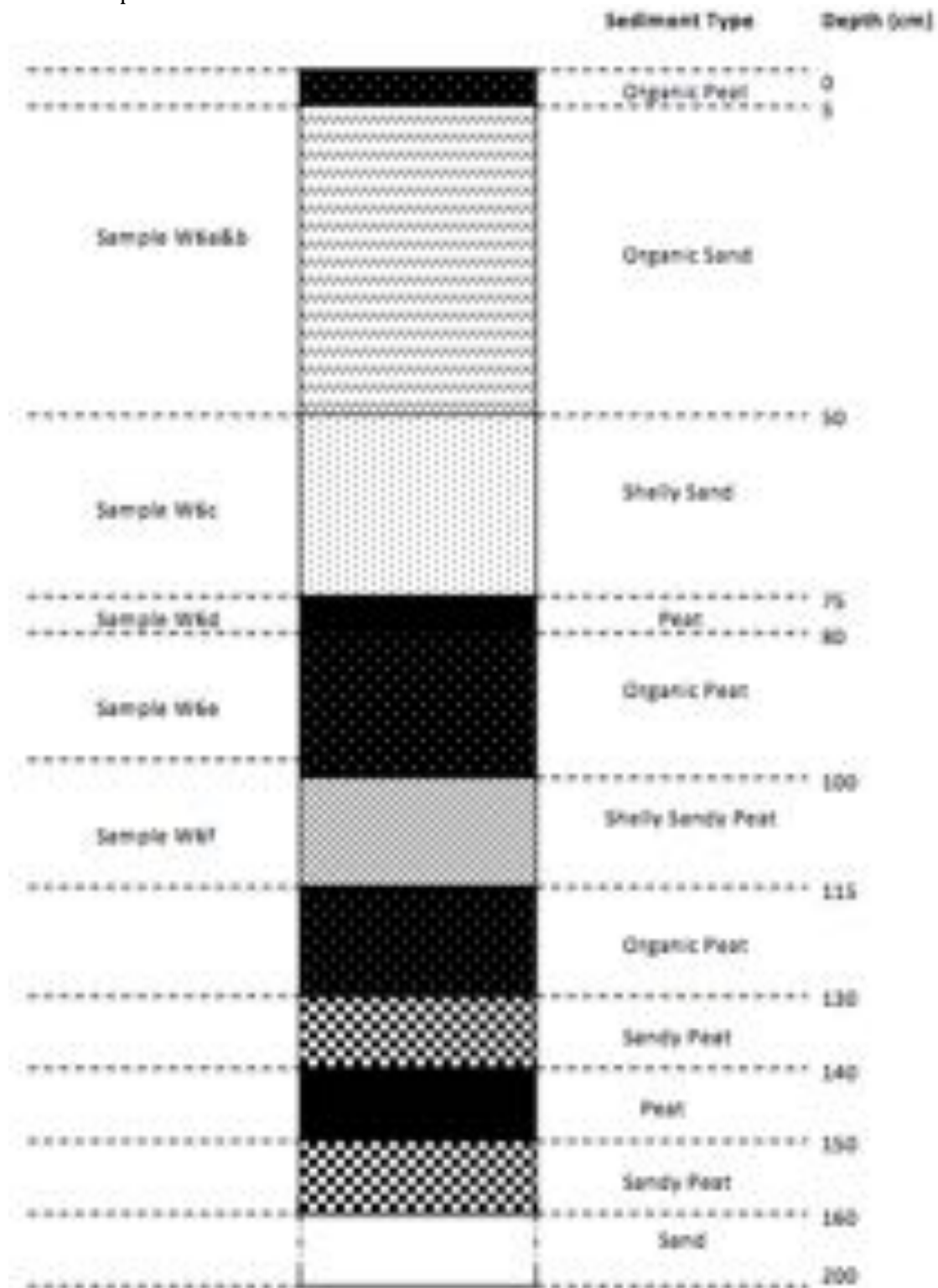
Sheskinmore Sediment Core W6

Coordinates: 54.807034, -8.469426

Surface Elevation (mOD): 3.23

Date Excavated: 17th June 2012

No. of Samples Recovered: 6



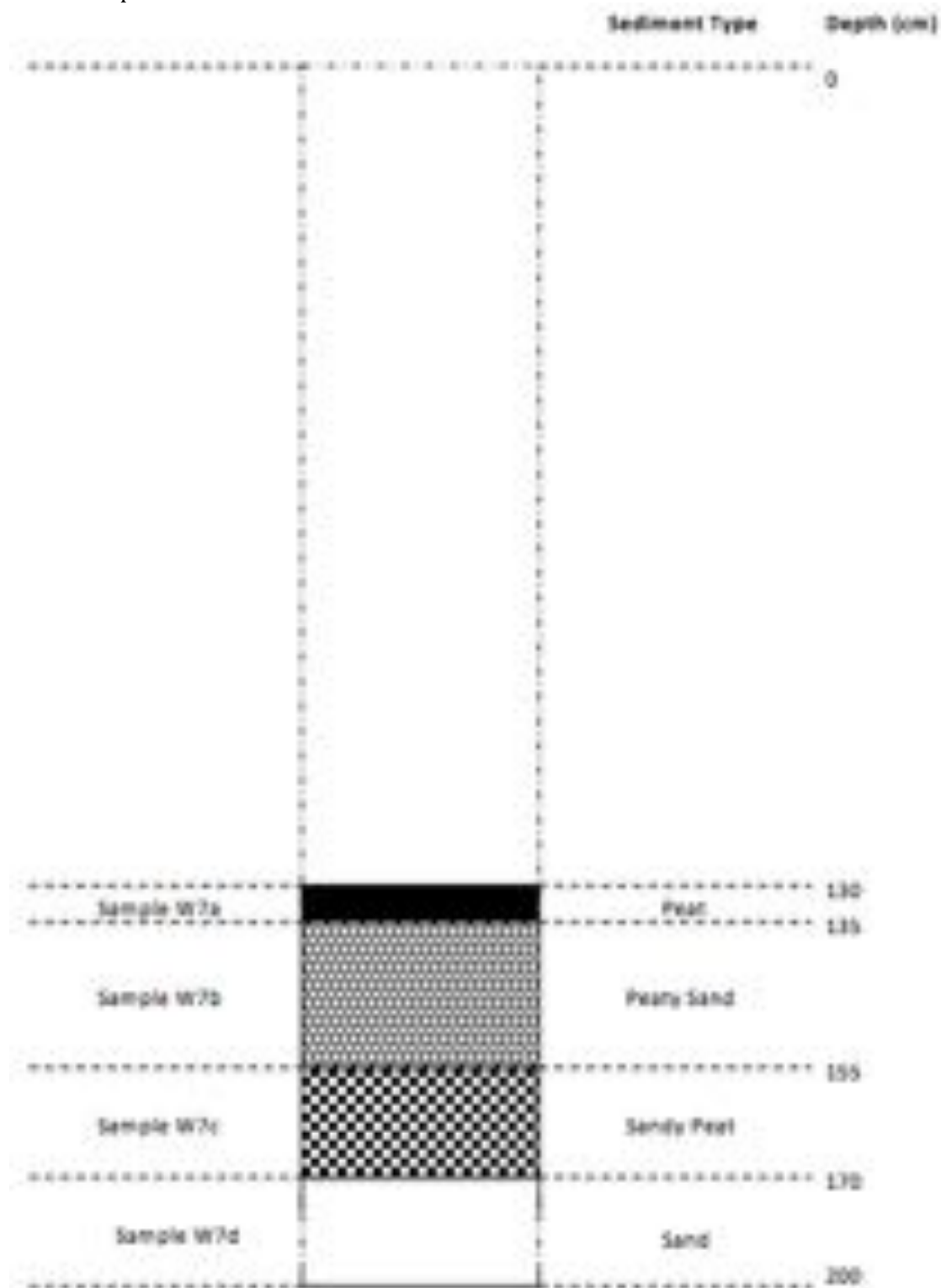
Sheskinmore Sediment Core W7

Coordinates: 54.806815, -8.464588

Surface Elevation (mOD): 3.21

Date Excavated: 17th June 2012

No. of Samples Recovered: 6



Appendix F. Sediment information for the two complete lake cores including: location, date excavated, number of samples recovered, and detailed cross sectional sedimentology with depth below ground surface.

Sheskinmore Lake Core 1

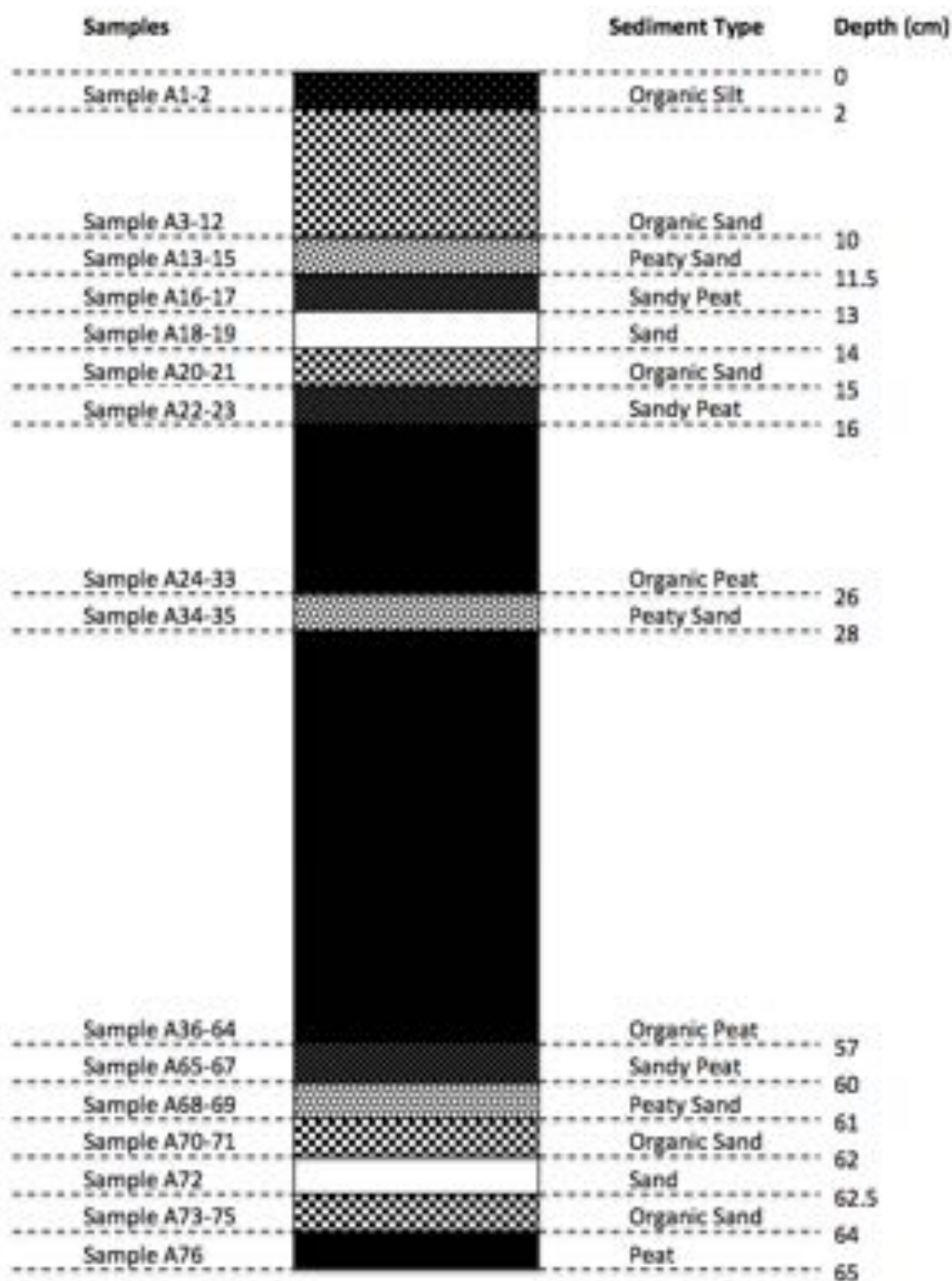
Coordinates: 54.80907205 -8.46424649

Surface Elevation: 2.62mOD

Date Excavated: 12th June 2013

Date Divided: 16th January 2014

No. of Samples Recovered: 76 samples from one 65cm length core



Sheskinmore Lake Core 2

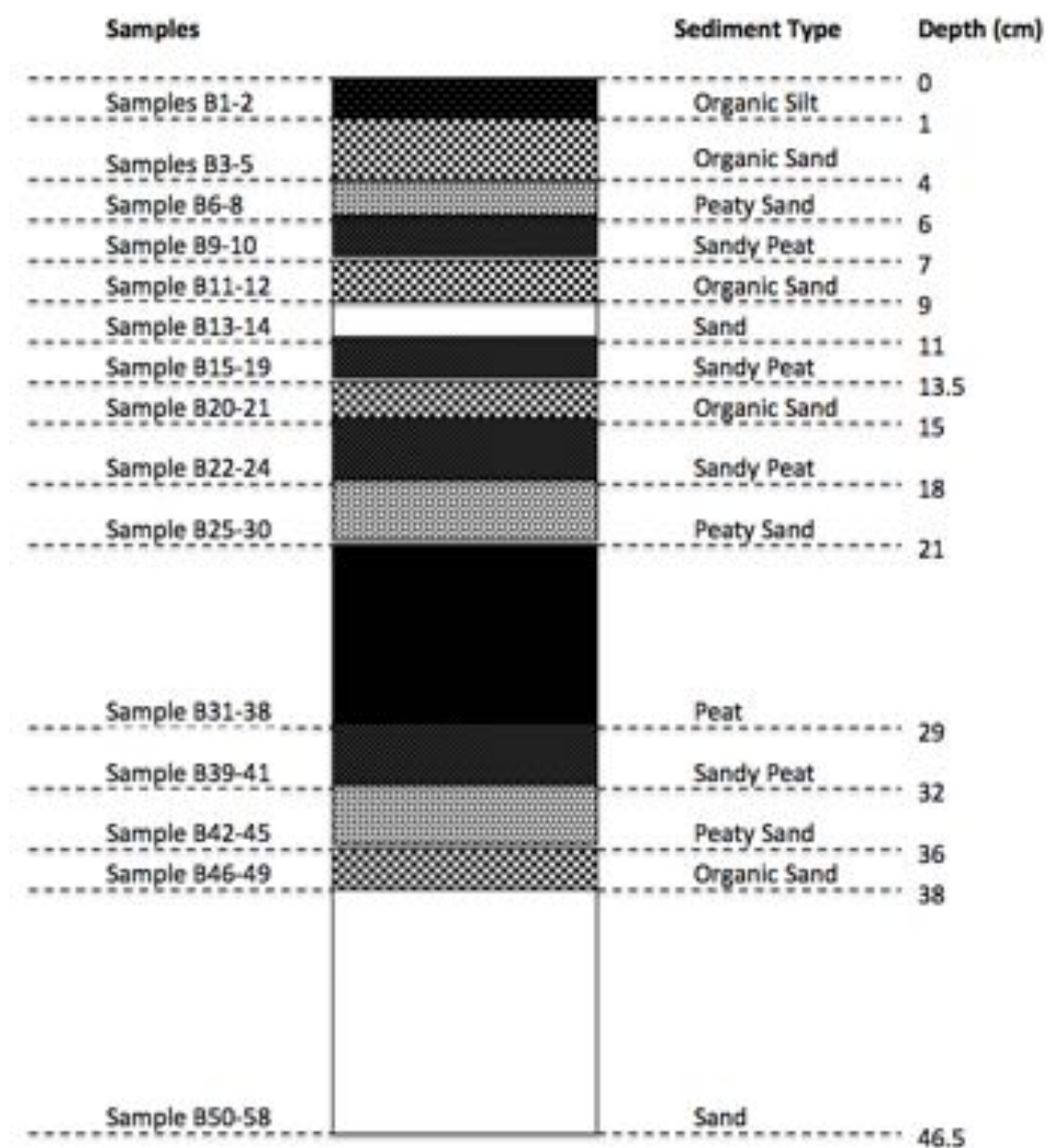
Coordinates: 54.80784989 -8.46509046

Surface Elevation: 2.90mOD

Date Excavated: 12th June 2013

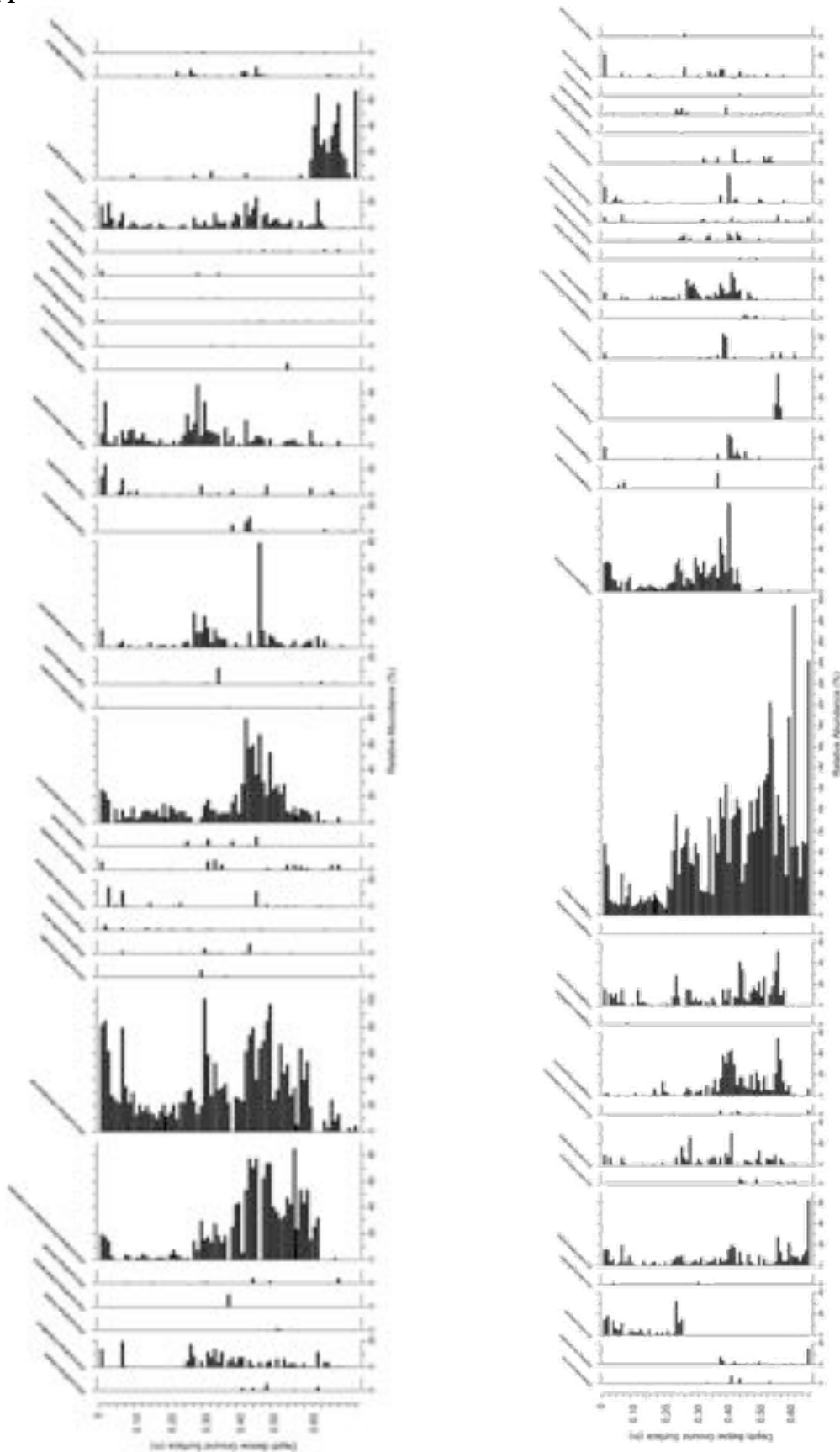
Date Divided: 16th January 2014

No. of Samples Recovered: 58 samples from one 46.5cm length core

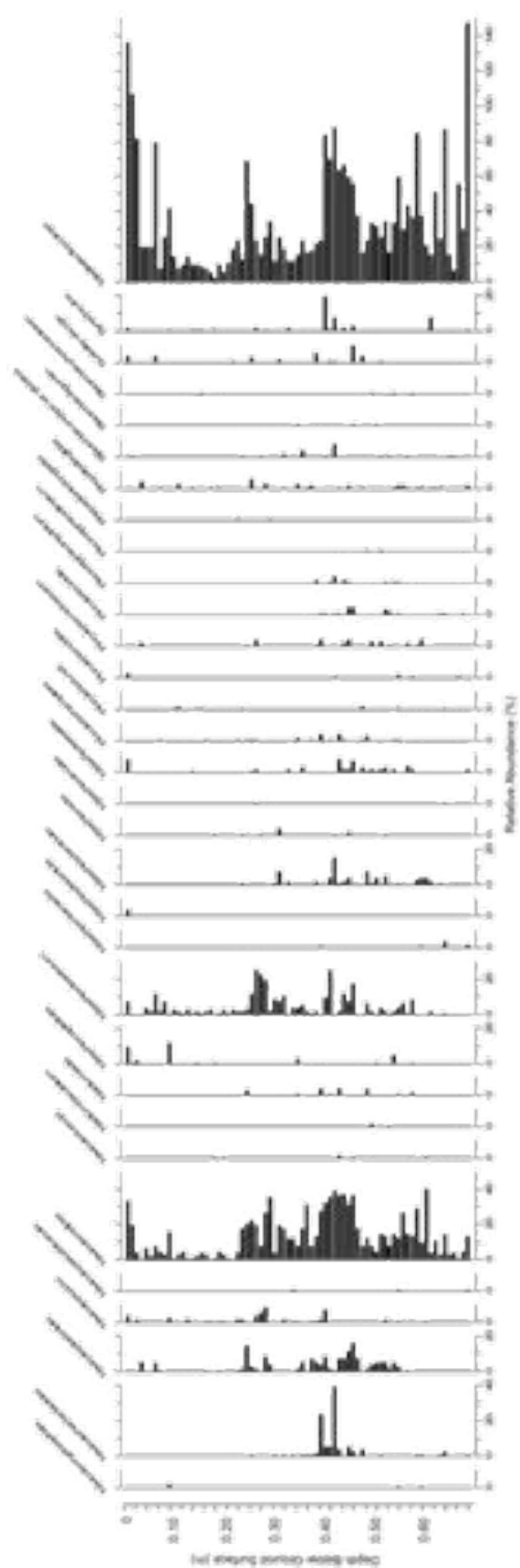


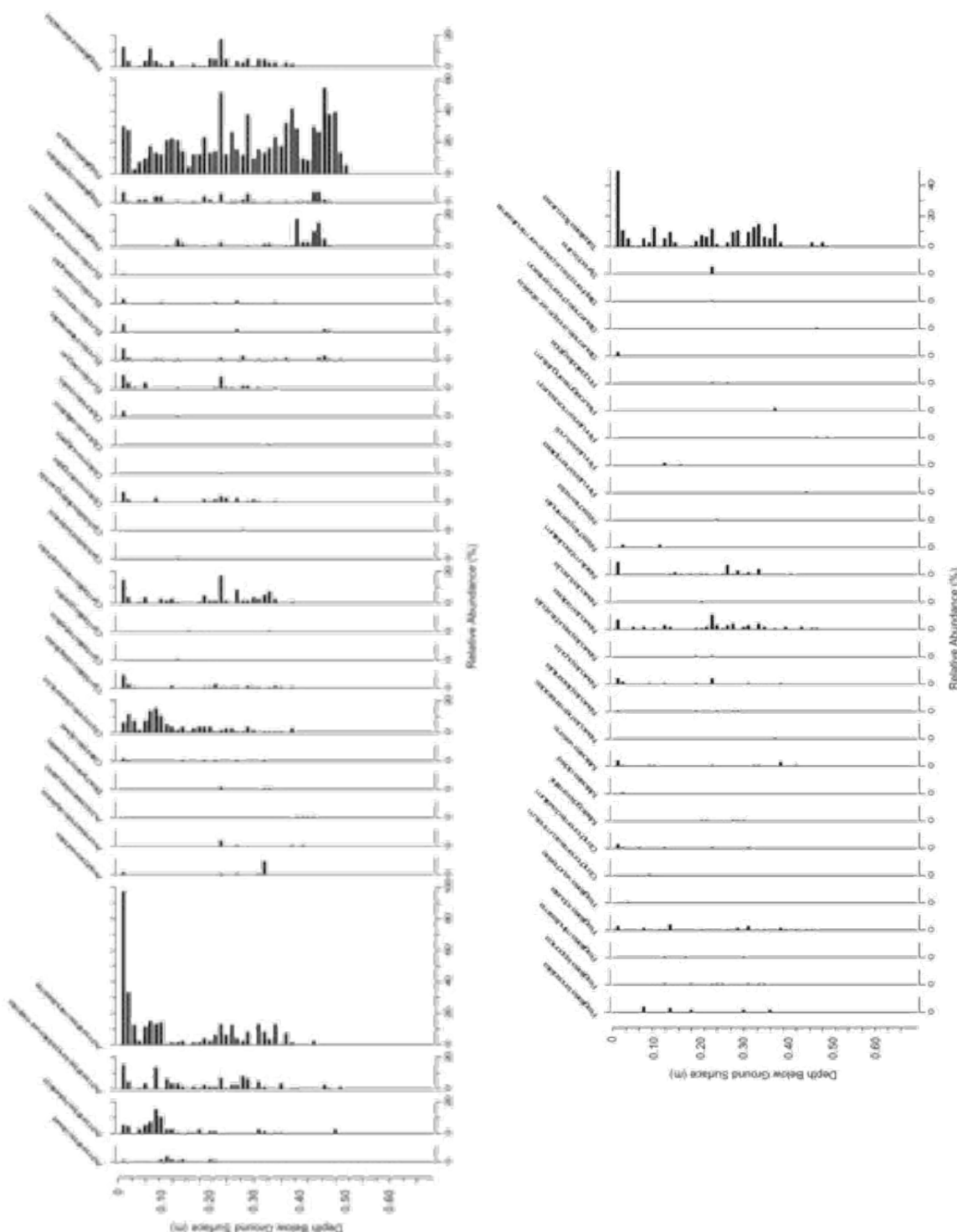
Appendix G. Down core variations in diatom percentage of all taxa identified in the two cores. Note stratigraphies are divided into three parts for Core 1 and two parts for Core 2.

Core 1

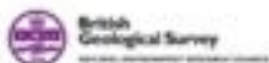


Core 1 continued...





Appendix H. Radiocarbon dating reports from the Natural Environment Research Council (NERC), East Kilbride, Scotland.



Radiocarbon Facility (East Kilbride)
 Sutton Enterprise Technology Park
 Rankine Avenue
 East Kilbride
 G75 0QF
 Email: N.C.Cisneros@nerc.uk.ac.uk
 Phone: +44 (0) 1295 260207
 Website: www.uk.ac.uk/nerc

25th February 2015

Dr H Burningham
 Department of Geography
 University College London
 Coastal & Estuarine Research Unit
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 London
 WC1E 6BT

Dear Helena

NRCF Radiocarbon Analytical Report 1791.0414.001

I enclose the formal report for two samples submitted under the above project number. The samples were prepared to graphite at the NERC Radiocarbon Facility-East Kilbride and passed to the SUERC AMS Laboratory for ^{14}C analysis. In keeping with international practice the results are reported as conventional radiocarbon years BP (relative to AD 1950) and $\delta^{13}\text{C}$, both expressed at the $\pm 1\sigma$ level for overall analytical confidence. Unless otherwise noted the results have been corrected to $\delta^{13}\text{C}_{\text{VPDB}} - 25$ using the $\delta^{13}\text{C}$ values provided in the report. The $\delta^{13}\text{C}$ values were measured on a dual inlet stable isotope mass spectrometer (Thermo Fisher Delta V) and are representative of $\delta^{13}\text{C}$ in the original, pre-treated sample material. The quoted precision is the uncertainty of repeated measurements of the same CO_2 aliquot, ie machine error only. Please let me know if you have any questions concerning the numerical significance of the results and/or the experimental procedures used.

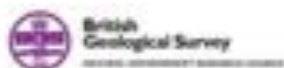
Publication is the prime indicator of science supported by NRCF and therefore an important means by which the value of the facility to the scientific community is judged. Please send us bibliographic reference to and/or copies of any publications (including those not peer-reviewed) and PhD theses, which discuss or describe the results in this report. If you have any comments on the significance of the results in the context of your original research objectives and/or on the report itself please also let us know. Such response will be fed back to the appropriate awarding Committee(s) at NERC. The above project allocation number should be quoted in any correspondence.

May I remind you that, to avoid ambiguity, the uniquely assigned lab code (SUERC-) and also the facilities where the samples were prepared to graphite and analysed should be quoted as an essential component in the publication/discussion of these data. Likewise, the support provided through NERC should be acknowledged in all relevant publications, to include the wording: This work was supported by the NERC Radiocarbon Facility NRCF010001 (allocation number 1791.0414). The fulfilment of this requirement will be considered in the allocation of future NERC support. For any appropriate accounting exercise the full economic cost of the two ^{14}C measurements is £1,280.

Best wishes

Dr Laz M Cisneros-Daarl

Enc. Radiocarbon Analytical Report



NERC Radiocarbon Facility (Environment)

RADIOCARBON ANALYTICAL REPORT

Allocation No: 1791.0414 **Submitter:** H. Burningham
University College London

Project Title: Meso-scale environmental forcing of coastal sedimentary lakes.

Sampling location: Sheskinmore Lough, County Donegal, Ireland

Sample composition: Sandy Peat

Pre-treatment of raw samples:

Samples were digested in 2M HCl (80°C, 8 hours), washed free from mineral acid with deionised water then digested in 1M KOH (80°C, 2 hours). The digestion was repeated using deionised water until no further humics were extracted. The residue was rinsed free of alkali, digested in 2M HCl (80°C, 5 hours) then rinsed free of acid, dried and homogenised. The total carbon in a known weight of the pre-treated sample was recovered as CO₂ by combustion with CuO in a sealed quartz tube. The gas was converted to graphite by Fe/Zn reduction.

Results

Publication Code	Sample Identifier	¹⁴ C Enrichment (% Modern ± 1σ)	Conventional Radiocarbon Age (years BP ± 1σ)	Carbon content (% by wt.)	δ ¹³ C _{org} ‰ ± 0.5
SUERC-57619	A67 59-60cm	95.71 ± 0.44	353 ± 37	4	-26.6
SUERC-57710	A17 12-13cm	65.50 ± 0.32	3398 ± 39	0.07	-26.6 (2)

Dr Luz M. Canerale-Dótal

25th February 2015

Notes:

- (1) The composition of both samples was predominantly sand, rather than peat and this can be observed in the pictures taken of the pre-treated samples (see below).
- (2) The % carbon contents of the samples were much lower than would be expected for peat and for sample combustion attempts were made to pick the organic material in preference to the sandy component.
- (3) ¹⁴C dates are not as expected in terms of estimated age and stratigraphic order. There were no problems during sample preparation and the quality control data were within acceptable limits. We suggest some potential reasons for the ¹⁴C dates obtained from the two samples submitted. Small dark fragments can be observed in both samples after pre-treatment (see pictures). These could be pieces of coal, which is resistant to the acid-alkali-acid pre-treatment and therefore would remain in the sample. The pre-treated sample photo for A17 shows a greater proportion of the dark fragments relative to the lighter organic material compared with the appearance of sample A67. This would enhance the shift to an older ¹⁴C age due to the presence of geologically old coal in the sample. One aspect to consider is the possibility of coastal coal inputs to the lake via aeolian deposition. Aeolian



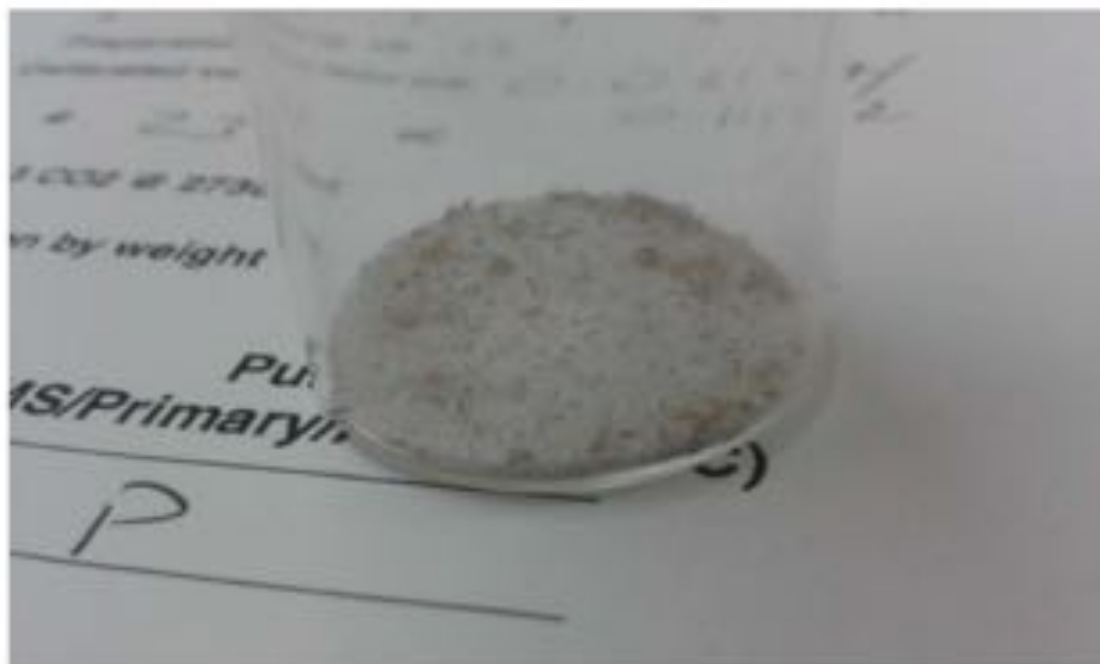
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sedimentation is likely at the study site as explained in your application and the layer beneath sample A17 in particular, is comprised of aeolian-derived sand as shown in Appendix 3.

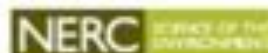
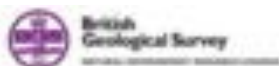
(4) We suggest sampling two other sections of the core from organic-rich layers.

Pre-treated sample A17



Pre-treated sample A67





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30th September 2015

Dr H Burningham
 Department of Geography
 University College London
 Coastal & Estuarine Research Unit
 Gower Street
 London
 WC1E 6BT

Dear Helene

NRCF Radiocarbon Analytical Report 1791.0414.002

I enclose the formal report for samples submitted under the above project number. The samples were prepared to graphite at the NERC Radiocarbon Facility-East Kilbride and passed to the SUERC AMS Laboratory for ^{14}C analysis. In keeping with international practice the results are reported as conventional radiocarbon years BP (relative to AD 1950) and % modern ^{14}C , both expressed at the $\pm 1\sigma$ level for overall analytical confidence. Unless otherwise noted the results have been corrected to $\delta^{13}\text{C}_{\text{VPDB}} = -25$ using the $\delta^{13}\text{C}$ values provided in the report. The $\delta^{13}\text{C}$ values were measured on a dual inlet stable isotope mass spectrometer (Thermo Fisher Delta V) and are representative of $\delta^{13}\text{C}$ in the original, pre-treated sample material. The quoted precision is the uncertainty of repeated measurements of the same CO_2 aliquot, ie machine error only. Please let me know if you have any questions concerning the numerical significance of the results and/or the experimental procedures used.

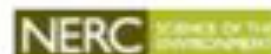
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Best wishes

Dr Luz M Cisneros-Deza

Enc. Radiocarbon Analytical Report



NERC Radiocarbon Facility (Environment)

RADIOCARBON ANALYTICAL REPORT

Allocation No: 1791.0414 **Submitter:** H Burningham
University College London

Project Title: Meso-scale environmental forcing of coastal sedimentary lakes.

Sampling location: Sheskinmore Lough, County Donegal, Ireland

Sample composition: Sandy Peat

Pre-treatment of raw sample:

Samples were digested in 2M HCl (80°C, 8 hours), washed free from mineral acid with deionised water then digested in 1M KOH (80°C, 2 hours). The digestion was repeated using deionised water until no further humics were extracted. The residue was rinsed free of alkali, digested in 2M HCl (80°C, 5 hours) then rinsed free of acid, dried and homogenised. The total carbon in a known weight of the pre-treated sample was recovered as CO₂ by combustion with CuO in a sealed quartz tube. The gas was converted to graphite by Fe/Zn reduction.

Results

Publication Code	Sample Identifier	¹⁴ C Enrichment (% Modern ± 1σ)	Conventional Radiocarbon Age (years BP ± 1σ)	Carbon content (% by wt.)	δ ¹³ C _{VPDB} ‰ ± 0.5
SUERC-61822	A25 17-18cm	92.42 ± 0.40	632 ± 35	2.2	-27.9
SUERC-61823	A63 55-56cm	96.25 ± 0.43	307 ± 35	8.9	-27.9

Dr Luz M. Cienfuegos-Dosal
30th September 2015